A comparative study of riparian drain management and its effects on phosphate and sediment inputs to Te Waihora/Lake Ellesmere.

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Abstract

Issues affecting water quality are seen as one of the most important and pressing global problems of our era. In New Zealand, water bodies with the poorest water quality and ecological condition tend to be surrounded by pastoral land use. Lake Ellesmere/Te Waihora in Canterbury, New Zealand, is a typical example of the issues that nutrient and sediment run-off from pastoral land can create.

The aim of this study was to determine the relationship between sediment concentrations, phosphate concentrations, ecological state and the degree of riparian restoration on drains that flowed into Lake Ellesmere/Te Waihora, and to calculate the load of phosphorus and sediment delivered by each of the drains to Te Waihora over the year, comparing this to the loads carried by larger, natural streams and rivers. Little research has been done on these small artificial tributaries of the Lake Ellesmere/Te Waihora catchment. Data collection was carried out on 10 drains with variable degrees of riparian planting, monthly in summer and autumn, and fortnightly in winter and spring, due to higher variability in drain flows during this time.

Sites 1, 2 had low dissolved oxygen (DO) and high total phosphorus (TP), lack of flow and extremely high conductivity, and (with) Site 5, higher suspended particulate matter (SPM) concentrations. All these factors are consistent with the lack of ecology occurring in these drains. All drains failed to meet the Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines for TP concentrations. All water chemistry parameters showed significant differences between seasons except conductivity. Mean

water temperatures and pH were higher in summer and lower in winter, while mean DO levels were higher in winter (and spring) and lower in summer (and autumn). Macroinvertebrate analyses indicated moderate to severe pollution in all the drains, despite the amount of riparian planting present and the presence of macroinvertebrate community structure was related mainly to substrate size.

The degree and type of riparian planting present on the drains studied did not appear to affect TP, SPM, macroinvertebrates or general water quality. This is likely to be due to the fact that little of the riparian planting had been specifically planted for restoration purposes. The highest loads of TP and SPM occurred in winter and spring, and in the larger (wider and deeper) drains. As flow increased in the drain, so did the load of phosphorus and sediment carried. Comparison with Environment Canterbury monitoring data for the river tributaries of the lake indicated that more TP and SPM is carried to the lake by natural rivers and streams, than by the drains, but the latter do make a significant contribution. The percentage of TP that is in dissolved form was higher than had previously been assumed, in both the drains and the larger, natural rivers and streams.

It is recommended that future restoration work aim to reduce the amount of phosphorus and sediment entering the larger drains in winter and spring. More adequate riparian planting needs to occur on these drains, and it needs to be managed in a way that a reduction in dissolved phosphorus levels is also achieved.

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Chapter 1: Introduction

1.1. Global issues for freshwater quality

Although the earth is frequently referred to as the 'blue planet', only 2.5% of the earth's water is freshwater (Oki and Kanae, 2006) and this includes glaciers, deep groundwaters and other inaccessible sources. With an increasing global population and increasing development (agriculture, urbanisation), negative impacts on our freshwater ecosystems are occurring at an alarming rate (Rockström and Karlberg, 2010; Prigent *et al.,* 2012). Competition for freshwater resources amongst human stakeholders has resulted in intense biodiversity declines that are far greater in freshwater ecosystems than those in the most affected terrestrial ecosystems (Dudgeon *et al*., 2006). Threats to aquatic ecosystems include water pollution, habitat degradation, species invasion, flow modification and over-exploitation of food species. However it is the issues of water quality that remain one of the most important and pressing global problems of our era to both human health and aquatic ecosystems (Strayer and Dudgeon, 2010; Onda *et al.,* 2012).

1.1.1. Nutrification of freshwaters

Lake eutrophication is the leading freshwater concern taking place both in developed and developing countries (Bricker and Devlin, 2011). Lake eutrophication can be defined as; an ecosystem response to the addition of artificial or natural substances to an aquatic system resulting in increased nutrients and changes to community composition (Smith and

Schindler, 2009). In lake ecosystems human induced eutrophication is also known as 'cultural eutrophication' and is currently considered one of the biggest issues facing most surface waters (Smith and Schindler, 2009). It has been shown that higher rates of eutrophication can coincide with economic development (Jin *et al*., 2006; Singh, 2008). Agriculture and urbanisation are the two main offenders of cultural eutrophication (Hall *et al*., 1999; Carpenter *et al*., 2007) and can have huge negative impacts on lake ecosystems. In lakes, eutrophication is associated with the degradation of ecosystem values and leads to algal blooms, hypoxia in the hypolimnion, fish kills and increased prevalence of toxic cyanobacteria (Nixon, 2009; Chuai *et al*., 2012; Salmaso and Cerasino, 2012).

Increased levels of phosphorous and nitrogen are seen as the main offenders of cultural eutrophication (Conley *et al*., 2009). These nutrients can be found in products such as fertilisers. Traditionally, nitrogen was seen as the main contributor to lake eutrophication (Dugan and McGauhey, 1974; Claesson and Ryding, 1975; Wilson and Sleigh, 1976). However, in some cases phosphorous is now identified as the main contributor (Carpenter, 2008; Schindler *et al*., 2008; Wang and Wang, 2009). Each lake needs to be evaluated on its own characteristics as it has been shown in New Zealand that both nitrogen and phosphorous can be limiting nutrients, where there is not an appropriate balance of the two nutrients occuring (Abell *et al*., 2010). The negative impacts of fertilisers on our freshwater ecosystems has been thoroughly studied (Chien *et al*., 2011; Gaxiola *et al*., 2011; Schindler *et al*., 2012). Run-off from agricultural practices results in higher nutrient loads entering our waterways. Point source pollution (identifiable from a single point, for example from a pipe or sewer) was the main contributor of pollution (excess nutrients, untreated sewage and storm water) to our freshwater ecosystems. However, recently, a majority of point source issues have been addressed and diffuse pollution (non point pollution, such as polluted runoff) is now thought to be the main concern in terms of pollution. Diffuse pollution is much harder to regulate as there can be many different sources of contamination (Fu *et al.,* 2013). This trend is exacerbated by the low application levels of methods that minimise nutrient run-off into streams, such as riparian planting (Hutchins *et al*., 2010; Collins *et al*., 2012).

1.1.2. New Zealand freshwater nutrient issues

With a temperate climate that allows grazing nearly all year round, fertile soils and a low population to land ratio; farming in New Zealand is the economic back-bone of the country (Ballingall and Lattimore, 2004; Monoghan ad Muirhead, 2008) with the majority of exports from the agricultural sector (MacLeod and Moller, 2006) . Before the growth of pastoral agriculture in New Zealand, more than 80% of the land was forested. This has changed to <30% native forest cover, primarily due to the grazing of nearly 60 million sheep and cattle. Pastoral agriculture is the chief land-use in the middle and lower catchment areas of New Zealand streams and rivers (Allan, 2004).

Between 1993 and 2003 there was a significant expansion in the dairy farming sector in New Zealand with national dairy cow numbers increasing by 44% (Houlbrooke *et al*., 2004). The intensification and diversification of the New Zealand agriculture industry in the past few decades has resulted in increased use of farm fertiliser and pesticides (MacLeod and Moller, 2006). Irrigation occurs on 500,000 hectares of land in New Zealand. 350,000 hectares are in Canterbury alone (Woods and Howard-Williams, 2004). As a result, many of New Zealand's low-land freshwater ecosystems are degraded or are at risk of becoming degraded due to nutrient leaching and run-off. Pastoral agriculture is the biggest source of water pollution in New Zealand (Wilcock, 1986; Ballentine and Davies-Colley, 2009; Verburg *et al*., 2010). A study conducted by the National Institute of Water and Atmospheric Research (NIWA), looking at water quality within New Zealand, showed lakes with the poorest water quality and ecological condition tend to be surrounded by pastoral land use (Verburg *et al*., 2010). The report used the Trophic Level Index (TLI). The TLI is used to describe lake water quality and to determine changes in the nutrient (trophic) status of lakes. Included is nitrogen and phosphorus levels, algal biomass and water clarity. In this report it stated “the TLI score increased with increasing percentage pastoral land cover and decreased with increasing percentage native or alpine land cover” and “lakes with pastoral land cover had poor ecological condition or were not vegetated”. The same has been shown with rivers where both nitrogen and phosphate concentration levels are increasing significantly, over time, at the national scale (Ballantine and Davies-Colley, 2009).

1.2. Reducing nutrient effects on freshwaters

Restoration work on croplands is a concept that has received global attention (Foley *et al*., 2005). New Zealand is no exception with calls for better farming practices or best management practices (BMPs) to be implemented across the country (Monoghan *et al*., 2007). Such practices intended to reduce the nutrient impacts on waterways include; riparian planting along waterways, putting fences up to exclude stock from waterways, focusing on certain areas (such as effluent ponds) and better irrigation techniques (Müller *et al*., 2010; McDowell and Campbell, 2011; McDowell and Nash, 2012). The understanding of

BMPs are needed to ensure New Zealand keeps its "clean and green" image which is an essential part of New Zealand's economy (Collins et al., 2010).

1.2.1 Riparian Planting

Riparian zones (the interface between land and a river, stream or lake) are very diverse and provide environmental services such as filtering nutrients, stabilising the microclimate of waterways, lowering sedimentation and erosion, sourcing allochthonous inputs into the ecosystem and creating habitat for wildlife (Naiman and Decamps, 1997). Riparian planting can be used as a conservation tool to help improve water quality in the following ways.

Filtering nutrients and lowering sedimentation

Excess loading of nutrients such as nitrogen and phosphorous in lakes often originates from tributaries (i.e streams and rivers flowing into the lake). The vegetation that is planted in the riparian zones needs nitrogen and phosphorus to grow (Vance, 2001). Vegetation can absorb nutrients through their roots before the excess nutrients enter streams and other water bodies.

Stream bank erosion is not only a waste of valuable soil assets but it also releases sediment and nutrients to waterways. Stream bank erosion is one of the major contributors of sediment into waterways (Wilkinson *et al.,* 2009). Increased levels of sedimentation in our waterways can have negative impacts such as lowering water clarity, covering habitat, increased levels of nutrients (nitrogen and phosphorus which can be bound to sediment), and blocking light penetration in the water column that allows macrophytes and algae to grow (Gillingham and Thorrold, 2000; Davies-Colley and Smith, 2001). Vegetation in the riparian zone, especially on stream banks, can stabilise banks and can also retain sediment that is entering waterways from across the surrounding areas (i.e from gullies) (Zhou and Shangguan, 2008; Loades *et al*., 2010; Vigiak *et al*., 2011).

Stabilising the microclimate

Riparian planting can have a huge impact on stream water temperatures due to shading that the overhanging planting provides. Over a 600-900m stretch of riparian planting, water temperatures can decrease by 4°C (and increase by 4°C in the absence of planting) (Rutherford, 2004).

The more sunlight/solar radiation a waterway receives, the more diurnal change in dissolved oxygen (DO), temperature and pH will occur (McDowell and Wilcock, 2008). DO, water temperature and pH are all important factors in regards to stream health.

Sourcing allochthonous inputs

Riparian plants add course particulate organic matter (CPOM) into waterways. Upon entering a stream, CPOM acts as a surface area for biofilm establishment, thus increasing in stream productivity and demand for nutrients (Aldridge *et al*., 2009). The reintroduction of CPOM from the riparian planting can increase microbial activity and phosphorus retention due to higher surface area availability. CPOM may provide pathways of energy inputs into food webs, thus increasing biological diversity and stability in addition to retaining more phosphorus to fuel further growth (Aldridge *et al*., 2009). This will minimise the load of phosphate and sediment being transported downstream into coastal water bodies.

Creating habitat for wildlife

Riparian planting is also a great conservation tool for maintaining regional diversity (Naiman *et al.*, 1993). Riparian zones provide food, habitats for predators and prey and predator avoidance for both terrestrial and freshwater organisms. For example aquatic organisms often need the structure provided from riparian vegetation at various stages of their life cycles (Post *et al.*, 2007). Examples of this include the New Zealand Dobsonfly (*Archichauliodes diversus*) that lays its eggs on surfaces (i.e leaves and branches) that are overhanging water (McLellan, 1975) and whitebait eggs are laid in moist bank vegetation (Richardson and Taylor, 2002).

1.2.2 Nutrient limit setting

Guidelines for maximum nutrient concentrations allowed in aquatic ecosystems and for human health protection are extremely important. Unfortunately, at a national scale, dissolved inorganic nitrogen and dissolved reactive phosphorous concentration often exceed recommended guidelines (Larned *et al.*, 2004). In New Zealand the ANZECC guidelines are usually applied and their overall stated objective is "to provide an authoritative guide for setting water quality objectives required to sustain current or likely future environmental values for natural and semi-natural water resources in Australia and New Zealand." With the variability of water bodies in New Zealand however, there has been criticism that their guidelines or nutrient limits cannot be applied in every stream or river. Guidelines based on average conditions may be too lenient for some sites but too strict for others (Larned *et al.*, 2004) . This highlights the importance of monitoring nutrient effects.

1.3 Monitoring nutrient effects

It is important to monitor the water quality of tributaries to see how the lake is changing and/or degrading. Important aspects to monitor include water flow, nutrient enrichment, suspended particulate matter, water chemistry (DO, temperature, pH and conductivity) and ecological state. When monitoring water quality parameters, it is vital to keep in mind that each parameter is only one piece of the puzzle and all aspects need to be considered.

1.3.1. Water flow

Water flow is essential for calculating nutrient and sediment loads in water (Howard-Williams and Pickmere, 2010). It is vital to understand the transport of these pollutants to effectively manage the loads entering downstream water bodies such as lakes and the coastal zone (Alexander *et al*., 2002). Streams with higher velocities and larger flows can transport greater amounts of nutrients and sediment.

Water flow, or discharge, is the volume of water that moves over a designated point over a fixed period of time. The flow of a tributary is related to the amount of water running off the surrounding watershed. It is affected by weather, increasing during rainstorms and decreasing during dry periods. It also changes seasonally, with lower flows during the summer months when evaporation rates are high (Allan and Castillo, 2007). Non climatic changes to water flow can occur due to anthropogenic disturbances (Milly *et al*., 2005), with a major contributor being the abstraction of water for irrigation of crops/pasture.
Freshwater ecosystems are strongly influenced by water flow. Many freshwater species are adapted to certain types of habitat which are related to flow and a river's flow regime is considered a 'master variable' that can drive variation (i.e the ability to process excess nutrients) within a freshwater ecosystem (Richter *et al*., 2003).

1.3.2 Nutrient and Suspended Particulate Matter (SPM) concentrations

Excess nutrients, as previously mentioned, can result in the eutrophication of lakes. It is necessary to monitor nutrient enrichment in tributaries to ensure pollution sources are known. Higher fluxes of total phosphorus and nitrogen run-off occur after the re-wetting of dry soils (Schönbrunner *et al.*, 2012). This is due to higher erosion rates (which can include sediment bound phosphorus) and run-off. Dissolved reactive phosphorus (DRP) and dissolved inorganic nitrogen (DIN) are more detrimental to our waterways as they are readily available for uptake by nuisance algae.

SPM concentrations in tributaries are important to monitor to calculate sediment loads entering downstream water bodies. Suspended matter in water columns can cause lack of light penetration from increased turbidity. This can have many impacts on the freshwater ecosystems. For example, macrophyte growth is generally light-limited, so a general decrease of submergent species is explained by changes in water transparency (Egertson *et al.*, 2004). Also, many species rely on visual cues for behaviour such as predation and communication. This can be heavily affected by increased turbidity.

1.3.3. Other quality parameters

Dissolved oxygen

Dissolved oxygen is a measure of the amount of oxygen dissolved in water. It is vital for freshwater fauna as many cannot survive under anoxic conditions. They need oxygen for respiration and for numerous other biological functions. Higher stream temperatures cause reductions in dissolved oxygen concentrations (Wilcock *et al.*, 1998; Morrill *et al.*, 2005) because temperature inversely controls the solubility of oxygen in water; as temperature increases, oxygen is less soluble.

Water temperature

Water temperature controls the rate that chemical, physical and biochemical processes can occur, such as the processing of nutrients. At lower temperatures processes are slower and at higher temperatures processes occur at an accelerated rate. High water temperatures can also have direct negative impacts on the health of freshwater flora and fauna. Many freshwater species cannot survive over a certain temperature, with anything over 20°C generally being seen as the limit for many New Zealand aquatic invertebrates to survive (Quinn *et al.*, 2004). Lower water temperatures can help decrease the growth of filamentous nuisance algae (Parkyn *et al.*, 2003). Temperature has even been shown to be the over-riding factor in controlling periphyton biomass, when compared with nutrients (N, P and N+P) in some systems (Mosisch *et al.*, 2001). Diurnal changes in water temperature will naturally occur as solar radiation plays a huge role, with temperatures higher during the day and lower at night (particularly in shallow stretches of rivers).

pH

pH is a measure of the acidic or basic (alkaline) nature of a solution. The concentration of the hydrogen (H^+) ion activity in a solution determines the pH. pH values are on a scale that ranges from 0 to 14. Values lower than 6 (more H^+ activity than OH^- activity) are considered acidic. Values higher than 8 (more OH^- activity than H^+ activity) are considered alkaline. Values between 6 and 8 are considered neutral. Aquatic life forms have adapted to certain pH levels. When these levels are not optimal, populations can decline and deaths occur (Baker and Schofield, 1982). Only a few species of algae can live in very low pH levels as they have to live with a limited supply of carbon dioxide, which is essential for photosynthesis (Gross, 2000). Low pH levels in waterways can also inhibit microbial metabolism which can affect the breakdown of CPOM (Chamier, 1987). On the other hand, high levels of photosynthesis can raise pH in water, and this leads to diurnal variables in pH.

Conductivity

Conductivity is a measure of the total ionic strength of the water. It measures a solution's ability to carry an electric current. It is widely used in water quality studies as a quick field indication of the level of ionic enrichment (i.e. saltwater ions, nutrient content) of the water. The higher the dissolved ion concentration, the more conductive the sample. Conductivity levels are influenced by many factors including; surrounding geology and soils, surrounding land-use, temperature and flow conditions. Dissolved salts are necessary for many freshwater fauna to survive, however high levels of salinity in freshwater ecosystems will cause the loss of many sensitive species, as they are not adapted to saline environments.

1.3.4 Ecological State

Ecological state of a waterway provides a measure of the cumulative effects of all pressures on a waterbody. The use of macroinvertebrate sampling as an indicator of ecological stream health is used both internationally and within New Zealand. Macroinvertebrates are often surveyed as part of water quality testing as the type and number of individuals found in an area can be related directly to water quality (Stark *et al*., 2001) . Various taxa differ in their habitat requirements and pollution tolerances and they respond predictably to these disturbances, which make them excellent indicators of the life supporting capacity of freshwater ecosystems (Hickey and Clements, 1998).

Macroinvertebrates are a diverse range of animals without backbones, usually defined as those that can be retained in a 0.5mm net or sieve. New Zealand has over 200 species of identified freshwater macroinvertebrates. They live in a range of environments, from lowland streams to the alpine headwaters. Macroinvertebrates are important links within stream foodwebs, between primary producers (including detrital inputs) and higher trophic levels, most notably fish (Wallace and Webster, 1996) and birds. While some studies have found no significant relationship between macroinvertebrate richness and percentage of stream riparian planting (Moore and Palmer, 2005), other studies have highlighted the importance of these buffers for macroinvertebrate presence (Watzin and McIntosh, 1999; Stewart *et al*., 2001). The overhanging vegetation keeps water temperatures constant and cool whilst also providing habitat for macroinvertebrates that complete their life cycle, as adults, on land and CPOM.

Macroinvertebrate communities can be heavily impacted by the nutrient levels and flow regimes of lakes and rivers. Increased algae, due to high nutrients levels, have been shown to increase the number of grazers such as Chironomidae and Emphemeroptera (Kiffney *et al*., 2001). Streams that suffer from high levels of pollution (i.e urban and pastoral streams) therefore tend to see a change in the macroinvertebrate community composition, with pollution tolerant taxa become more dominant (Hall *et al*., 2001). Low flows can see macroinvertebrate communities become less diverse, as less water means a loss of habitat and loss of food sources or other important interactions (Dewson *et al*., 2009).

Diversity indices and biotic indices can be used to assess macroinvertebrate conditions (see methods). Macroinvertebrate assemblage can be impacted by many factors highlighting the importance of multiple stressors (Townsend *et al*., 2008; Matthaei *et al*., 2010). This is why macroinvertebrates are good indicators for stream health; they give a broader picture of stream health including both physical and chemical factors. For example, invertebrate densities have been found to be lower when both fine sediment levels are higher and flow rates are lower (Matthaei *et al*., 2010). Benthic substrate is also an important factor with macroinvertebrates preferring stony bottomed streams to silt/mud and impervious surfaces (Barnes *et al*., 2013; Hopkins and Olson, 2013).

Other biotic indices, such as fish, can sometimes be used to determine water quality; however in New Zealand the large number of diadromous fish greatly affects the abundance and presence of fish in a riverine system (McDowall, 1993) which can confound results if the species is not present at the time of sampling. The implications of sampling fish species for monitoring stream health can also be problematic, especially where electro-fishing is required. Specific training and expensive equipment is necessary.

1.4 Nutrient issues in Lake Ellesmere/Te Waihora

Lake Ellesmere/Te Waihora is no exception to many of the issues that face lakes, both on the national and international stage. The lake has been recognised as a wetland of international importance for wildlife as it is one of the few large, coastal, brackish lagoon habitats found in New Zealand. It is home to many endemic, endangered and migratory species (O'Donnell, 1985). Unfortunately in 2010, it was also named the second most polluted lake in New Zealand in terms of nutrient content and algal growth (Verburg *et al.*, 2010).

1.4.1. Physical characteristics of the lake

Lake Ellesmere/Te Waihora (LE/TW) is a situated west of Banks Peninsula in Canterbury New Zealand, and is separated from the Pacific Ocean by the Kaitorete Spit (Figure 1.1). The lake covers 20,000 hectares, making it New Zealand's fifth largest lake, with approximately 75km of shoreline. The narrow spit which separates the lake from the sea is manually opened to the sea periodically to control the level of the lake and to ensure that the surrounding agricultural land is not inundated with water (Gough and Ward, 1996).

Fig. 1.1: Location of Lake Ellesmere/Te Waihora and Kaitorete Spit in relation to Christchurch, New Zealand.

LE/TW has many farm drains flowing into it and has over 40 tributary drains, streams and rivers in its catchment, which drains a total of 256,000 hectares (Collins *et al.*, 2012). The lake has undergone many changes since it was formed less than 5,000 years ago (Kirk and Lauder, 2000). It is currently a brackish lagoon with an average depth of 1.4metres. The catchement climate is relatively dry, receiving an average annual rainfall of 500 – 750mm with the Southern Alps, combined with westerly winds that blow across New Zealand, providing a rainshadow effect (Renwick et al., 2010). Before the arrival of Maori (approximately 800 years ago) the catchment was forested in podocarps. Today over 80% of the catchment has been converted into agricultural pasture (Hughey and Taylor, 2008) which has increased nutrient transport to the lake.

1.4.2. Lake Ecology

The shore vegetation surrounding Lake Ellesmere/Te Waihora is a mixture of native and exotic with the abundance of native brackish species increasing and the number of native freshwater wetland species decreasing (Hughey and Taylor, 2008). Within Lake Ellesmere/Te Waihora, macrophyte numbers have always fluctuated (Gerbeaux, 1989) although they have never diminished to the level they are now. A large disturbance in 1968 (Wahine storm) caused a huge loss of macrophytes as the severe winds ripped them out of the lake bed. As a by-product, the lake crossed an ecological threshold from a clear-water regime to a turbid one. Macrophytes in shallow lakes stabilise sediments and reduce the cycling of phosphorous to phytoplankton (Søndergaard *et al*., 2003). When macrophytes are lost, as in the case of Lake Ellesmere, winds (water mixing) can cause sediment to remain suspended in the water column and the phosphorus is utilised by phytoplankton (algae). This results in turbidity and algal blooms. As a consequence macrophytes are unable to re-grow due to shading effects (Folke *et al*., 2004). A study by Gerbeaux (1993) looked at the possibility of trying to re-introduce two macrophyte plants (*Lepilaena bilocularis* and *Ruppia polycarpa*) back into the lake. It concluded that the best time to try and re-introduce the plants would be when lake levels are low enabling more light penetration to reach the lake bed. Better water quality was needed however to limit phytoplankton growth.

There are 16 species of fish found in the lower catchment of the lake and 10 in the upper catchments (Environment Canterbury (ECan), 2011). The higher number in the lower catchment is most likely due to the diadromous fish that spend part of their life cycle in the marine environment.

There are many threatened fish species or species that are in gradual decline in the lake and within its catchment. For example, the Canterbury mudfish (*Neochanna burrowsius*) is listed as a nationally critical species meaning they are highly threatened with extinction (Allibone *et al.,* 2010) but can be found in the LE/TW catchment (ECan, 2011). The key reason for their decline within the lake has been put down to the drainage of key wetland habitat. Also, shortfinned eels (*Anguilla australis*) are abundant in the lake but they are getting smaller (but not younger) (Jellyman and Todd, 1998). The reduction in size does not seem to be related to the over harvesting of this species, but rather the changing environment of the lake itself (i.e loss of macrophytes and changing macroinvertebrate communities).

The birdlife at LE/TW is seen as internationally important due to the high diversity that can be found there (O'Donnell, 2000). The bird species found in the catchment cover a wide range of guilds. Invertebrate and fish species found in the catchment are seen as an important factor contributing towards bird presence in and around the lake.

1.4.3. Human uses of the lakes resources

Human use of the lake involves many recreational activities. These activities include camping, fishing, bird-watching, cycling, water sports and hunting. For example, LE/TW is New Zealand's most popular recreational duck shooting area. It is regarded as a very important resource in the community by many different groups.

Lake Ellesmere/Te Waihora is culturally significant to the local Māori tribe Ngāi Tahu. The natural resources at Te Waihora are deeply important to the tribe which has inhabited the area for over 40 generations. Te Waihora or 'water spread out' was also known historically as Te Kete Ika a Rākaihautū (the fishing basket of Rākaihautū). This name signifies the lakes traditional importance as a food source (mahinga kai). Lowered water quality and habitat degradation has resulted in the scarcity of many species that were once abundant in the lake making the mahinga kai value negligible (Hearnshaw and Hughey, 2010). Māori have a view that it is vital to ensure natural resources and the environment is protected. This is achieved through a concept called Kaitiakitanga, which means guardianship and ensuring resources are used sustainably to guarantee future generation's use (Roberts *et al*., 1995). Ngāi Tahu has Kaitiakitanga over Te Waihora. Although there are continuous debates surrounding lake management, regarding outcomes that will be beneficial for all stakeholders, Ngāi Tahu are currently in a Joint Management Plan with the Department of Conservation (DOC) to enhance the lake's significant values. Other schemes include the Whakaroa Te Waihora cultural and ecological restoration programme led by Ngāi Tahu and ECan as well as the co-governance arrangement included in the Canterbury Water Management Strategy between Ngāi Tahu and the Canterbury Regional Council for the active management of LE/TW and its catchment.

The Waihora Ellesmere Trust (WET) is a community organisation committed to the restoration and enhancement of Lake Ellesmere. One of their main objectives is to "promote and support riparian plantings" in the catchment (Varona, 2010).

1.4.4. Nutrient and sediment levels in the lake

A study conducted by NIWA looked at the state and trends of water quality in New Zealand lakes with LE/TW as one of the study sites (Verburg *et al*., 2010). It found that LE/TW was very heavily impacted (in terms of nitrogen, phosphorus and suspended sediment concentrations) by its surrounding land cover type (pastoral).

Lake clarity, measured by Secchi disk, has shown a constant decline in recent years (Hughey *et al*., 2013) due to increasing phytoplankton levels, suspended sediments in the lake and sediment inputs. This in turn is due to increasing nutrient levels, wind driven re-suspension and lake edge erosion. Nitrogen and phosphate levels in the lake usually (> 90% of the time) exceed the concentrations needed by phytoplankton (Larned and Schallenburg, 2006). Dissolved phosphorus concentrations have increased in all lake tributary sites monitored between 1993 - 2007 while dissolved nitrogen has decreased (Hughey *et al*., 2013). Also, Phosphorus concentrations recorded in the lake were generally higher than other coastal lakes around New Zealand. Due to the shallowness of the lake, wind induced turbulence of lake bed sediments results in sediment bound phosphorus input remaining in suspension in the water (Larned and Schallenburg, 2006). The continued input of phosphorus from the catchment adds to what is already in the lake sediment, and is a major concern.

1.4.5. Sources of phosphorus entering the lake

Phosphorus contributions have been estimated as 90% from tributaries, 6% from rainfall and 4% from birdlife (e.g waterfowl) (Hughey and Taylor, 2008). It has been suggested in the past by Environment Canterbury (ECan), that agricultural drains and smaller tributaries of the lake, cumulatively provide much of the external phosphorus load into the lake. While there have been studies that focus on the phosphorus contribution of larger natural streams and rivers in the catchment, little work has been done on the agricultural drains/smaller tributaries. Larned and Schallenburg (2006) did not include flow data for drains and tributaries (especially the smaller tributaries) or high frequency and spatial coverage of nutrient sampling, or sampling of storm flows on tributaries. It also remains unclear whether riparian planting affects the amount of phosphorus and SPM carried in the drains. A previous study looking at the effectiveness of the riparian zones (Collins *et al.*, 2012) on phosphorus and nitrogen levels in four separate streams/creeks. Although results showed significant increases in DO and conductivity and significant decreases in turbidity in planted sites, no other differences were found between planted and non-planted sites.

1.5 Research aim and objectives

The aim of this study is to quantify the phosphorus and SPM loads entering LE/TW from agricultural drains and assess whether riparian planting on the drains effectively reduce phosphorus and sediment loads, or improve ecosystem health. These results will be compared to the loads entering from the larger, natural rivers. This research will expand information on the flow of smaller tributaries flowing into the lake, and on phosphorus and SPM loads flowing into the lake, including during heavy rainfall events. It will also provide more evidence of the effect of riparian planting on phosphorus and SPM loads, ecological state and general water chemistry.

This will be achieved through the following objectives:

- Selecting farm drains around the shores of LE/TW which are representative of different degrees of riparian planting in their catchments
- Routine sampling (monthly in summer and autumn, fortnightly in winter and spring) and analysis of water in the selected representative drains for; flow, phosphorus concentrations, SPM, temperature, dissolved oxygen, pH, conductivity and ecological state through invertebrate sampling.
- Determining the degree of correlation between SPM and phosphorus concentrations, ecological state and the degree of riparian restoration on the drains.
- Calculating the load of phosphorus and SPM delivered by each of the drains to LE/TW over the year, and comparing this to the load carried by larger, natural rivers over this time
- Make recommendations on how to reduce the load of phosphorus and SPM coming from the drains.

Chapter 2: Methods

This chapter outlines the methodology used to assess phosphorus and SPM inputs into LE/TW and the effects of riparian management on drain phosphorus, SPM, water quality and invertebrate health.

2.1. Study design:

The location of Lake Ellesmere/Te Waihora can be seen in Figure 2.1.

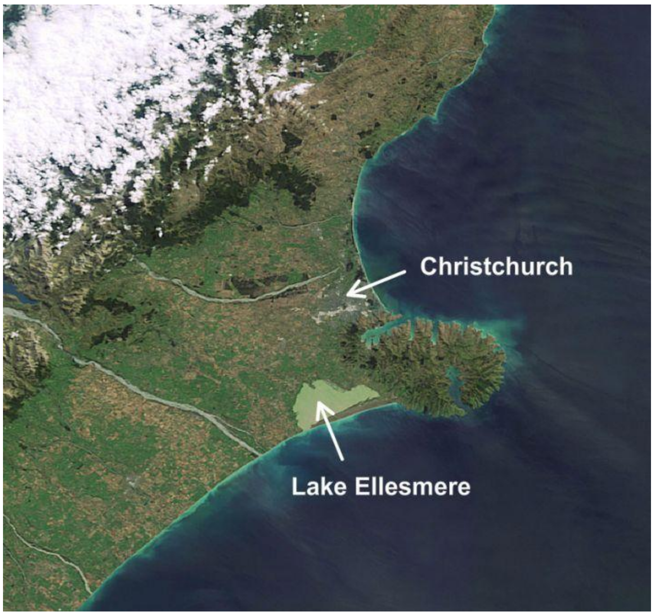

Fig. 2.1: Map showing the location of Lake Ellesmere/Te Waihora in relation to Christchurch, New Zealand.

2.1.1. Site selection

Ten agricultural drains flowing into LE/TW, with varying degrees of riparian planting, were selected in this study. The locations of the 10 sites can be seen in Figure 2.2. For this study the definition of an agricultural drain was a manmade outlet that ran along or through some type of agriculture landuse, in which water was flowing, in this case, toward LE/TW. Selection criteria for the drains were as follows:

- An accessible location close to the lake, but not affected by the water backflow
- variability in the percentage of riparian cover between the drains
- represenatative of local drain sizes

Sites were chosen as close to the lake as possible, without getting backflow from the lake, in order to determine the load of phosphorus and SPM entering the lake. Drains were selected whilst ensuring there was varying amounts of riparian cover between them (more on this in section 2.1.2). Accessibility to the drains was also taken into consideration for both time management, and health and safety reasons.

Fig. 2.2: Sampling locations of the 10 drain sites around Lake Ellesmere/Te Waihora, New Zealand.

2.1.2. Assessing riparian planting

The selected drains needed to have varying percentages of riparian planting to see whether or not the degree of riparian planting made any difference to the water quality entering the lake.

To calculate the percentage of riparian planting on each drain, Google Earth software and groundtruthing, was used. Goggle Earth was used in January 2013 to view potential sites. Then using the ruler tool on google earth, a 1000m upstream interval upstream of the sampling site was marked. Figure 2.3 shows sites 6-9, with the 1000m interval marked out.

Fig.2.3: Google map image of sites 6-9 with the 1000m interval upstream of sampling site marked in red (Lake Ellesmere/Te Waihora, New Zealand).

The percentage of that 1000m inverval with some form of riparian planting on it was then calculated. Because drains ran alongside agricultural land, many of the sites had roads running along one side. This made riparian planting along both sides impossible, but nutrient inflows would only come from one side. Therefore, when calculating the percentage of riparian cover, if the drain ran alongside a road the percentage was calculated over 1000m. If the drain ran through a farming area where planting could occur on both sides, the percentage was calculated over 2000m. For example, Figure 2.4 shows a Google Earth image of site 10. The red line represents the 1000m upstream of the sampling site. The yellow line represents the location of any riparian planting along that 1000m. The ruler tool indicates the yellow line is 126m long. There is no road running alongside this drain so planting was possible on both sides. Therefore the calculation is 126/2000 = 6.3% (126m of bank out of a possible 2000m of bank is covered by planting).

Fig. 2.4: Goggle Earth image showing the 1000m interval upstream of the sampling site for site 10 (redline), and the measurement of how much of that drain had some form of riparian planting (yellow line).

Due to the drains running alongside farms, much of the riparian cover was in the form of shelter belts. From here on, this is referred to as "accidental cover". Although not riparian planting that has purposely been planted for environmental restoration, it is planting within the riparian margin and can still carry out the functions of reducing erosion and minimising pollutant run-off into the drains.

2.1.3. Sampling frequency and duration

A 12 month sampling period was chosen to identify seasonal differences and to get accurate annual phosphorus and sediment loads entering the lake via the drains. Water samples for phosphorus (total and dissolved), SPM and flow were collected at each site, monthly in summer and autumn due to the lower rainfall and fortnightly in winter and spring to obtain greater variability and gain representative results of this time. Invertebrate sampling was undertaken at each site, monthly throughout the year. Samples were collected during the day, and sites visited in the same order each time to try to minimise the effect of diurnal variation.

2.2. Site Descriptions

A table showing sites with data collection GPS co-ordinates, percentage of riparian cover and substrate type can be found in Table 2.1 (substrate type was recorded for each drain to enable the correct macroinvertebrate sampling protocols were used). Substrate types/sediment size were; cobble (64 – 255 mm), gravel (2 – 63 mm), silt/sand/mud (< 2 mm) and impervious (e.g concrete). These were measured once during the sampling period.

Site	GPS of data collection site	Percentage of riparian cover	Substrate type found in drain
1	43°43’33.86”S 172°30’40.48”E	5	Silt/Sand/Mud
2	43°43’10.13”S 172°25’45.49”E	45	Silt/Sand/Mud
3	43°41’35.75”S 172°26’32.69”E	0	Silt/Gravel
4	43°41’45.22”S 172°24’59.81”E	42	Gravel/Silt
5	43°43’47.04”S 172°23’19.49”E	7.5	Silt/Impervious
6	43°45’6.47”S 172°22’18.19”E	94	Cobble
7	43°45’59.54”S 172°22’7.87”E	86	Gravel/Silt
8	43°46’13.54”S 172°21’10.29”E	27	Cobble
9	43°46’58.00”S 172°20’21.90”E	0	Cobble
10	43°49’36.71”S 172°21’8.75”E	6.3	Cobble

Table 2.1: Data for; GPS co-ordinates, percentage of riparian cover and substrate type for each sampling site.

2.2.1. Site 1 (Clarks Rd)

Fig. 2.5: Sampling site location at site 1 (May 2013) looking downstream.

Fig. 2.6: Upstream view at site 1 (August 2013).

Photos of the sampling site and an upstream view of the drain at site 1 can be seen in figures 2.5 and 2.6. The riparian cover on this site is possible only on one side of the drain as there is a road running along the run of the drain. From using both Google Earth imagery and ground truthing there is 5% riparian cover on this drain. The riparian cover is “accidental”. The drain bed at this site is mostly silt/sand/mud (S/S/M). Land use along this drain is primarily pasture for cows. Cows are able to access the drain, with manure and hoof marks regularly seen in and around the drain at the sampling site, and cows were also seen in the drain. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.2. Site 2 (Embankment Rd.)

Fig. 2.7: Sampling site location at Site 2 (April 2013) looking downstream.

Fig. 2.8: Upstream view at site 2 (April 2013)

Photos of the sampling site and an upstream view of the drain at site 2 can be seen in figures 2.7 and 2.8. The riparian cover on this site is possible only on one side of the drain as there is a road running along the run of the drain. From using both Google Earth imagery and ground truthing there is 45% riparian cover on this drain. The riparian cover is "accidental" near the sampling location but does have some purposefully planted riparian cover upstream. The drain bed at this site is mostly S/S/M. Land use along this drain is primarily pasture for cows but there is sheep pasture as well. Both cows and sheep are able to access the drain, with manure and hoof marks regularly seen in and around the drain at the sampling site, and cows were also observed in the drain. In July, a sheep was found, dead, in the drain near the sampling site. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.3. Site 3 (Powells/Pannetts Rd)

Fig. 2.9: Upstream view at site 3. (April 2013).

Fig. 2.10: Sampling site location at site 3 (April 2013).

Photos of the sampling site and an upstream view of the drain at site 3 can be seen in Figures 2.9 and 2.10. The riparian cover on this site is possible only on one side of the drain as there is a road running along the run of the drain. From using both Google Earth imagery and ground truthing there is 0% riparian cover on this drain. A shelter belt was planted near the sampling site in April (Figure 2.9). The drain bed at this site was mostly cobble until the banks were weeded in April, and the drain became mostly silt/gravel (S/G). It did not recover by the end of sampling. Land use along this drain is pasture for cows and deer and fences keep these animals out of the drain. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.4. Site 4 (Near Coes Ford)

Fig. 2.11: Upstream photo of site 4 (April 2013).

Fig. 2.12: Sampling site location at site 4. (April 2013).

Photos of the sampling site and an upstream view of the drain at site 4 can be seen in Figures 2.11 and 2.12. The riparian cover on this site is possible on both sides of the drain. From using both Google Earth imagery and ground truthing there is 42% riparian cover on this drain. The riparian cover is both “accidental” (upstream) and purposefully planted (sampling point). There is fencing along the drain. The drain bed at this site is mostly S/G. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.5. Site 5 (Wolf Creek)

Fig. 2.13: Sampling site location at site 5 (May 2013).

Fig. 2.14: Upstream photo of site 5 (June 2013).

Photos of the sampling site and an upstream view of the drain at site 5 can be seen in Figures 2.13 and 2.14. The riparian cover on this site is possible on both sides of the drain. From using both Google Earth imagery and ground truthing there is 7.5% riparian cover on this drain. The riparian cover is both "accidental". Animals have access to the drain with sheep observed in the drain during sampling. A sheep was found, dead, in the drain February 2013.The drain bed at this site is mostly S/S/M and impervious (concrete). The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.6. Site 6 (Hanmer Rd.)

Fig. 2.15: Sampling site location at site 6 (April 2013).

Fig. 2.16: Upstream photo of site 6 (June 2013).

Photos of the sampling site and an upstream view of the drain at site 6 can be seen in Figures 2.15 and 2.16. The riparian cover on this site is possible on one side of the drain. From using both Google Earth imagery and ground truthing there is 94% riparian cover on this drain. The riparian cover is “accidental”. There is fencing along the drain. Land use along the drain is primarily cow pasture. The drain bed at this site is mostly cobble (C). The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.7. Site 7 (Colletts Rd)

Fig. 2.17: Sampling site location at site 7 (April 2013).

Fig. 2.18: Upstream photo at site 7 (April 2013).

Photos of the sampling site and an upstream view of the drain at site 7 can be seen in Figures 2.17 and 2.18. The riparian cover on this site is possible on one side of the drain. From using both Google Earth imagery and ground truthing there is 86% riparian cover on this drain. The riparian cover is purposefully planted. There is fencing along the drain. Land use along the drain is primarily cow pasture. The drain bed at this site is mostly silt/gravel The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.8. Site 8 (Drain Rd.)

Fig. 2.19: data collection location at site 8 (June 2013).

Fig.2.20: Upstream photo of site 8 (May 2013).

Photos of the data collection location and an upstream photo of site 8 can be seen in Figures 2.19 and 2.20. The riparian cover on this site is possible on one side of the drain. From using both Google Earth imagery and ground truthing there is 27% riparian cover on this drain. The riparian cover is “accidental”. There is fencing along the drain. Land use along the drain is primarily cow pasture. The drain bed at this site is mostly cobble. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.9. Site 9 (Tramway Rd.)

Fig. 2.21: Data collection location at site 9 (May 2013).

Fig. 2.22: Upstream photo of site 9 (May 2013).

Photos of the data collection location and an upstream photo of site 9 can be seen in Figures 2.21 and 2.22. The riparian cover on this site is possible on one side of the drain. From using both Google Earth imagery and ground truthing there is 0% riparian cover on this drain. There is fencing along the drain. Land use along the drain is primarily cow pasture. Irrigation was seen directly entering the drain during data collection in March and April 2013. The drain bed at this site is mostly cobble. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.2.10. Site 10 (off Lower Lake Rd.)

Fig. 2.23: Data collection location at site 10 (May 2013).

Fig. 2.24: Upstream photo of site 10 (May 2013).

Photos of the data collection location and an upstream photo of site 10 can be seen in Figures 2.23 and 2.24. The riparian cover on this site is possible on one side of the drain. From using both Google Earth imagery and ground truthing there is 6.3% riparian cover on this drain. There is fencing along the drain. Land use along the drain is primarily deer pasture. The drain bed at this site is mostly cobble. The sampling site was chosen where the road allows easy access to the site, and there is ease of access in and out of the drain.

2.3 Sample collection and analysis

2.3.1. Water quality

On every sampling date, at each site; dissolved oxygen, temperature, conductivity and pH were measured. A HACH HQ40d multi meter with attached CDC401 conductivity, luminescent dissolved oxygen and a Shindengen ISFET pH probe was used. All probes were calibrated prior to fieldwork.

2.3.2. Invertebrate sampling

While species richness determines how many species are found in the dataset of interest, the use of diversity indices alone has fallen out of favour in the literature. Too much information is lost when trying to summarise the information gathered (Environment, 1999). The use of species richness alone is not recommended (Lenant, 1988) although this information is gathered to obtain the necessary data for other macroinvertebrate evaluations.

The generalisation of the taxa scores is valid for contaminants which are better associated with nutrient and organic enrichment (Quinn and Hickey, 1990). In the past, the Macroinvertebrate Community Index (MCI) and its quantitative alternatives have received poor performance reviews believed to be caused by incorrect tolerance scores for taxa when looking at heavy metal contamination (Hickey and Clements, 1998). It also has to be highlighted that the MCI methodologies were designed for use in stony streams and may not perform adequately in other habitats.

Invertebrate samples were collected from each drain that had water, once a month. Invertebrate samples were collected using semi-quantitative sampling protocols outlined in “protocols for sampling macroinvertebrates in wadeable streams” (Stark *et al*., 2001). The sampling methods used depended on the site’s bed characteristics. These methods were:

i) Protocol C1 – for hard-bottomed, semi quantitative sampling. This was used in drains that had gravely beds. The sampler wore waders and used a triangular-frame net. An area of habitat was chosen and the net was placed on the bed. The sampler then stood slightly upstream of the net, and the foot-kick method (Frost *et al*., 1971) was used. Substrate was disturbed by kicking (within 0.5 m from the net mouth) which was collected in the net. Each collection gathered approximately 0.3m² of streambed. This protocol was repeated at two other areas of habitat in each drain to include approximately 0.9m² of streambed at each site (Stark *et al*., 2001).

ii) Protocol C2- for soft-bottomed, semi quantitative sampling. This was used in drains that had silt/mud or macrophyte dominated beds. It is the soft-bottom equivalent to protocol C1 sampling. The sampler wore waders and used a triangular-frame net. At each site, three main habitats (outlined by Stark et al, 2001) bank margins, woody debris and aquatic macrophytes were sampled. In bank margins and aquatic macrophytes, substrate was jabbed aggressively with the net over an area of about 1m². The net was then swept through the substrate twice to collect dislodged organisms. Woody debris was picked up and any organisms were dislodged by hand into the net. Each of these techniques collects around 0.3m² of streambed. Ten habitats were chosen in each drain to include approximately 3m² of streambed as recommended by Stark *et al.* (2001). More area is needed in this protocol to gather the same range of invertebrates as protocol C1). During collection, the net was placed just above the bottom of the streambed to try and minimise the collection of fine detritus, sand, and mud.

At each site, samples were transferred to 250 mL sample containers with drain water, labelled and returned to the Waterways laboratory at Lincoln University. Once back at the lab, all samples were preserved in ethanol until identification could be carried out. Before identification, each sample was put through a sieve (0.5µm pore size) and the remaining organisms were transferred to a white tray, where they were identified under a microscope to the family or genus level using identification guides from Winterbourn *et al*. (2006).

2.3.3. SPM concentration

On every sampling date, SPM samples were taken from the drains that had flowing water (see section 2.3.5 for water flow measurement). A 1L water sample was collected, labelled and taken back to the laboratory where samples were pumped through a pre-weighed filter paper (0.45µm pore size) using a MV8010 mityvac pump and nalgene filtering apparatus. The filter paper was then air dried by placing it on a clean surface for 24 hours. The paper was then reweighed to determine the amount of SPM in the water in g/L. Detection limit was 0.0001 g/L.

2.3.4. Phosphorus analysis

To test the phosphorus concentration in the drain water, two 60mL samples of water were collected at each site (taking care not to disturb benthic sediment). One sample was filtered (using 0.45µm pore size filters and a syringe) in the field and the other was not filtered.

The filtered sample was analysed on site using the HACH DR3900 field spectrophotometer to determine the amount of dissolved phosphorus (as phosphate PO_4) using the 4500-P E ascorbic acid method (APHA, 1995) method. These PO_4 calculations were then converted to dissolved reactive phosphorus (PO_4 - P or DRP) by dividing by 3.07. Detection limit was 0.05 mg/L.

The unfiltered sample was used to determine total phosphorus (TP). This sample was labelled and placed into a chilly bin to be returned to the soil and water laboratory at Lincoln University and frozen until able to be analysed. Samples were analysed using the 4500-P B 5 persulfate digestion and the 4500-P E ascorbic acid method (APHA, 1995). Detection limits were 0.005 mg/L.

2.3.5. Flow

At each site, water flow data was collected (when possible) using a Global Water's FP111 flow probe. The probe was used by orienting the propeller directly into the flow, and reading the water velocity off the display (m/s). To get accurate readings of the velocity and stream channel morphology, the drains were divided into subsections, depending on the flow channels within the drain. For each subsection the width (w), depth (h) and velocity (f) parameters were measured (Figure 2.25). Using this information the flow of the drains was calculated as sum of flows in each subsection of the drain.

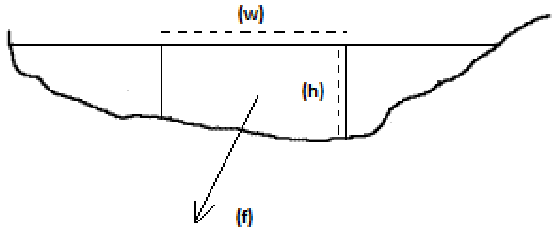

Fig. 2.25: The measurements taken for each subsection of drains at data collection locations to calculate flow parameters (w=width, h=depth and f=velocity).

2.4 Data Analysis

2.4.1 Macroinvertebrate analysis

After invertebrates were identified, the species richness, percentage Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, Macroinvertebrate Community Index (MCI) and Semi-quantitative Macroinvertebrate Community Index (SQMCI) were calculated. Species richness is calculated by counting the number of species (taxa) in your sample. EPT% is the calculation of how many taxa in the sample belong to the Ephemeroptera, Plecotera and Trichoptera orders. These 3 orders are considered to be very sensitive to water quality and have very low tolerances to pollution (Lenat, 1988). The more diversity in these 3 orders, the better the water quality. The MCI is a biotic index where prior allocations of scores are given to taxa based on their pollution tolerances. These scores are between 1 and 10, with a score of 10 indicating high sensitivity to pollution. These scores are then used to calculate the MCI score, by adding the taxon scores for invertebrates found in the sample and dividing that number by the total number of taxa, then multiplying this number by 20 (Figure 2.26). Scores will range between 0 (no species present) and 200 (when all taxa present have taxon scores of 10). Anything with a score above 120 is considered to be “pristine” (Stark, 1993).

$$\mathrm{MCI} = \frac{\sum_{i=1}^{i=S} a_i}{S} \times 20$$

Fig. 2.26: MCI calculation.

The Semi Quantitative Macroinvertebrate Community Index (SQMCI) uses the same taxon scores as the MCI but takes taxon abundance into account. Taxa are assessed by relative abundance rare, common, abundant, very abundant, very very abundant (Table 2.2).

Table 2.2. The abundance class, count numbers and coded abundance scores for the SQMCI biotic index calculation.

Abundance Class	Counts	Coded Abundance
Rare	1-4	1
Common	5-19	5
Abundant	20-99	20
Very abundant	100-499	100
Very very abundant	500+	500

The SQMCI score is determined by calculating the sum of the coded abundance scores multiplied by the taxon score for each taxa in a sample, divided by the total of the coded abundances for the whole sample (Figure 2.27). A score above 6 indicates clean water whereas a score of lower than 4 indicates severe pollution (Stark, 1998).

$$\mathrm{SQMCI} = \sum_{i=1}^{i=s}\frac{(n \times a)}{N}$$

Fig. 2.27: SQMCI calculation.

2.4.2 Flow data analysis

For analysis the total water flow (discharge) in litres/second (l/s) for each drain was calculated. Width in metres (w) x height in metres (h) yielded the area of the subsection in m^2. The area of the subsection was then multiplied by the velocity of water through that subsection, which was measured in m/s (Figure 2.25),giving a water flow in m^3/s.

All subsections were then added together to get the total flow or discharge of the drains in m^3/s. This calculation was then converted from m^3/s to L/s ($1m^3/s$ = 1000 L/s).

2.4.3 SPM data analysis

To calculate SPM loads delivered from the drains into the lake, the amount of SPM calculated in g/L was multiplied by the flow data collected for that particular drain at that time (g/s). These values were then multiplied up to kg/month and kg/year. Results were then used to make comparisons between the percentage of riparian cover and SPM loads and to calculate the annual SPM load input to the lake from the drains. This was then compared to the inputs from the larger, natural rivers and streams at this time (data provided by Ecan).

2.4.4 Phosphorus data analysis

To calculate TP loads delivered by the drains into the lake, the amount of total phosphorus (mg/L) was multiplied by the flow data collected for that particular drain at that time (mg/s). These results were then multiplied up to kg/month and kg/year and were used to make comparisons between the percentage of riparian cover and phosphorus loads. The data were also used to calculate the annual loading provided into the lake, which was compared to the inputs from the larger, natural rivers and streams over this time (data provided by Ecan).

DRP measured on site (mg/L) was used to determine how much of the TP was in dissolved form as PO_4. However at low TP concentrations (< 0.1 mg/L), measurements of field DRP were too unreliable to be compared with total phosphorus. Field DRP measurements of < 0.03 mg/L (close to or below detection) were not included in the comparison either.

2.4.5. Statistical analysis

All statistical analyses were performed in Excel (2007). Data was run through a linear regression model to determine if there was a relationship between factors (relationship between percentage of riparian planting and effects). Data was also run through a single-factor ANOVA to determine if there were significant differences between “groups” (i.e., seasons or site). Both tests were tested at α (type one error rate or false positive) 0.05. Tests were considered significant if the P value was less than 0.05 (the observed result would be highly unlikely under the null hypothesis). Degrees of freedom (d.f) is the number of values in the final calculation of a statistic that are free to vary. F is the F statistic used to help calculate the P value.

When an ANOVA output indicated significant differences ($P = <0.05$) , pairwise comparisons of means using Fishers least significant difference (LSD) tests ($\alpha = 0.05$) were undertaken to determine which “groups” had significant differences between them.

2.5 Limitations of current study

As with any study, human error during data collection and analysis is a potential limitation of this study. The methods for data collection used were kept constant across the field seasons and all analysis was carried out by the same person to try and minimise any variability.

Lack of water in the drains over summer was problematic. No data was collected in January 2013 for example as all of the drains had no water or no flowing water, due to the drought that plagued Canterbury at this time.

The field HACH spectrometer had limitations in detecting low levels of DRP phosphate (as <.05 mg/L) and the use of solid reagents interfered with light absorbtion, creating abnormally high results. These have not been used in the data analysis as they were higher than TP and clearly in error.

Flow data may be lower than it is in reality. During data collection, water could sometimes be seen flowing in the drains, when the flow was not strong enough to register on the flow meter. Also when flow rates were extremely high, and the safety of the data collector entering the drain was called in to question, stream morphology data was estimated from the bank (e.g site 4 in June, July and August).

The lack of purposely planted riparian zones close to the lake and the fact that shelter belts have been included as "accidental" riparian plantings in this study, may also be a limitation to this study. No research could be found that looks at the role of shelter belts in terms of removing nutrients, lowering sediment inputs and improving water quality.

Lastly, the estimations of the total load of phosphorus and SPM are rough estimates. Averages from the river data set from ECan and averages of the 10 drains sampled in this study were used to calculate the estimates. Google earth was used to count the number of drains entering LE/TW although the definition of a drain/stream and river can be interpreted differently.

Chapter 3: Results

3.1 Introduction

In this section, the results of sampling and statistical analysis are presented. Water chemistry, phosphorus, SPM, macroinvertebrates, flow then load data are presented.

For all seasonal analysis carried out; "Summer" refers to the months January, February and December, "Autumn" refers to the months March, April, May; "Winter" refers to the months June, July August and "Spring" refers to the months September, October and November.

When calculating loads of total phosphorus and SPM entering the lake, data from June was excluded. The data collected in this month was an outlier when carrying out statistical analysis. It has also been shown, that flow levels for June 2013 were much higher than June flows in previous years (Hughey *et al.*, 2013, Figure 5.2).

Note that some statistically non significant results have been included in this chapter, as these results still represent important findings, for example in assessing effectiveness of riparian planting on water quality.

Raw water chemistry and flow data can be found in the appendix.

3.2 General water chemistry

3.2.1. Dissolved oxygen (DO)

Across all sampling dates and all sites the lowest DO concentration was 0.2mg/L at site 2 and the highest was 14.1mg/L at site 8. All sites experience low DO concentrations (<5mg/L) on some occasions, but this was most pronounced at sites 1 and 2. The averages for each site ranged between 2.1 at site 2 and 10.1 at sites 8 and 6 (Table 3.1).

Table 3.1. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for dissolved oxygen (mg/L) at each sampling site over the entire sampling period.

Site	Mean	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	3.1	10	0.3 - 9.1	3.4	1.1
2 (Embankment Rd.)	2.1	10	0.2 – 6.5	2.2	0.7
3 (Powells/Pannetts Rd.)	8.4	11	2.7 – 12.7	3.8	1.1
4 (Near Coes Ford)	9.8	8	4.4 – 11.9	2.4	0.8
5 (Wolf Creek)	9.5	8	4.1 – 12.8	2.6	0.9
6 (Hanmer Rd.)	10.1	11	3.0 – 12.4	2.7	0.8
7 (Colletts Rd.)	6.1	11	3.3 - 9.4	2.1	0.6
8 (Drain Rd.)	10.1	11	4.6 – 14.1	2.9	0.9
9 (Tramway Rd.)	9.4	11	7.0 - 12.0	1.7	0.5
10 (Off Lower Lake Rd.)	9.4	11	5.9 - 11.2	1.5	0.5

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between mean dissolved oxygen concentration (mg/L) and the percentage of riparian planting on each drain. There was no significant relationship between dissolved oxygen and percentage of riparian planting along each site ($F= 0.01$, $d.f=9$, $p= 0.9$) (Figure 3.1).

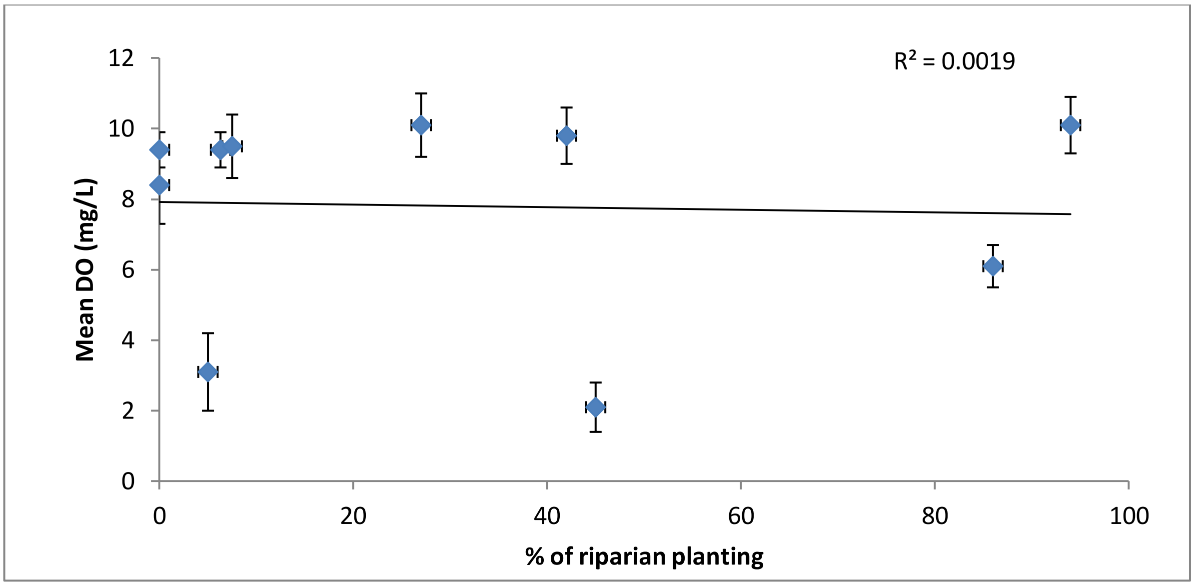

Fig. 3.1. The relationship between the mean dissolved oxygen concentrations (+/- S.E) and percentage of riparian planting for each site, across all sampling dates.

Seasonal variation effects

A single factor anova was carried out on all data to test for differences in DO (mg/L) levels across the seasons. There was a significant difference in DO (mg/L) levels between the seasons at α0.05 (F=7.0, d.f= 101, p= <0.001) (Figure 3.2). A least significance difference test was carried out to see which groups differed. At α0.05 significance, both winter and spring differed from autumn and summer. Summer and autumn did not differ from each other and spring and winter did not differ from each another.

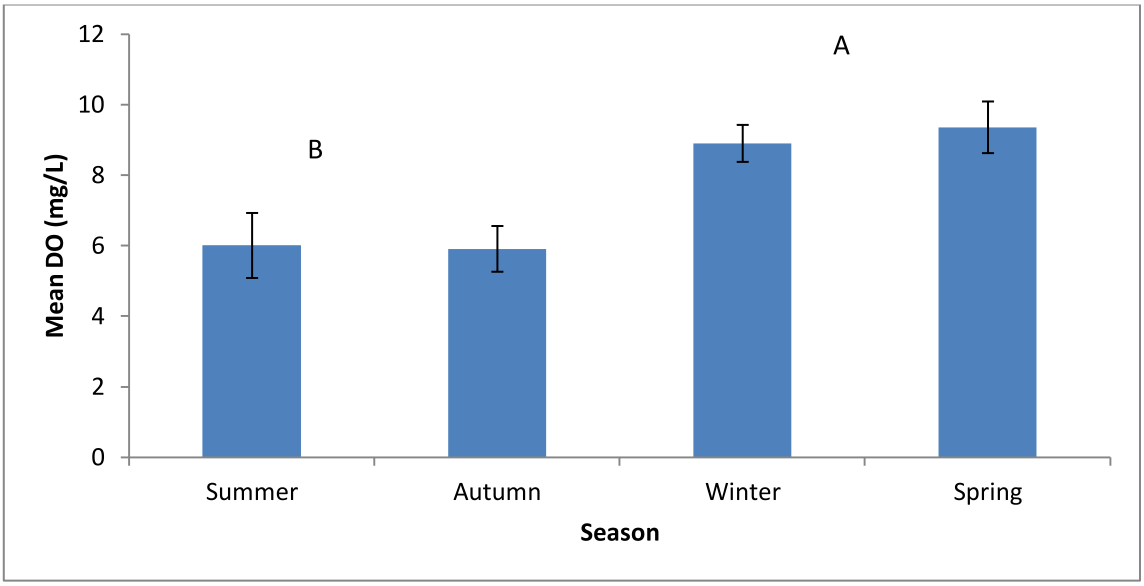

Fig. 3.2. Mean DO (mg/L) concentrations (+/- S.E) for each season across all sampling dates. Letters indicate significantly different levels at α0.05.

3.2.2. pH

Across all sampling dates and all sites the lowest pH recording was 6.5 at sites 7 and 8 and the highest was 9.1 at site 6. The averages for each site ranged between 6.9 at site 7 and 7.8 at site 6 (Table 3.2).

Table 3.2. Mean, number of samples (N), min. - max. values, standard deviation (S>D) and standard error (S.E.) for pH at each sampling site over the sampling dates.

Site	Mean	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	7.3	10	6.9 - 7.8	0.3	0.1
2 (Embankment Rd.)	7.4	10	6.9 - 8.4	0.6	0.2
3 (Powells/Pannetts Rd.)	7.2	11	6.6 - 8.6	0.7	0.2
4 (Near Coes Ford)	7.3	8	6.9 - 7.9	0.4	0.1
5 (Wolf Creek)	7.2	8	6.6 - 7.8	0.5	0.2
6 (Hanmer Rd.)	7.8	11	6.7 - 9.1	0.7	0.2
7 (Colletts Rd.)	6.9	11	6.5 - 7.3	0.3	0.1
8 (Drain Rd.)	7.5	11	6.5 - 8.4	0.7	0.2
9 (Tramway Rd.)	7.1	11	6.6 – 8.0	0.4	0.1
10 (Off Lower Lake Rd.)	7.2	11	6.6 - 7.9	0.4	0.1

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between mean pH and the percentage of riparian planting on each drain. There was no significant relationship between mean pH and percentage of riparian planting along each site (F= 0.89, d.f=9, p= 0.3) (Figure 3.1).

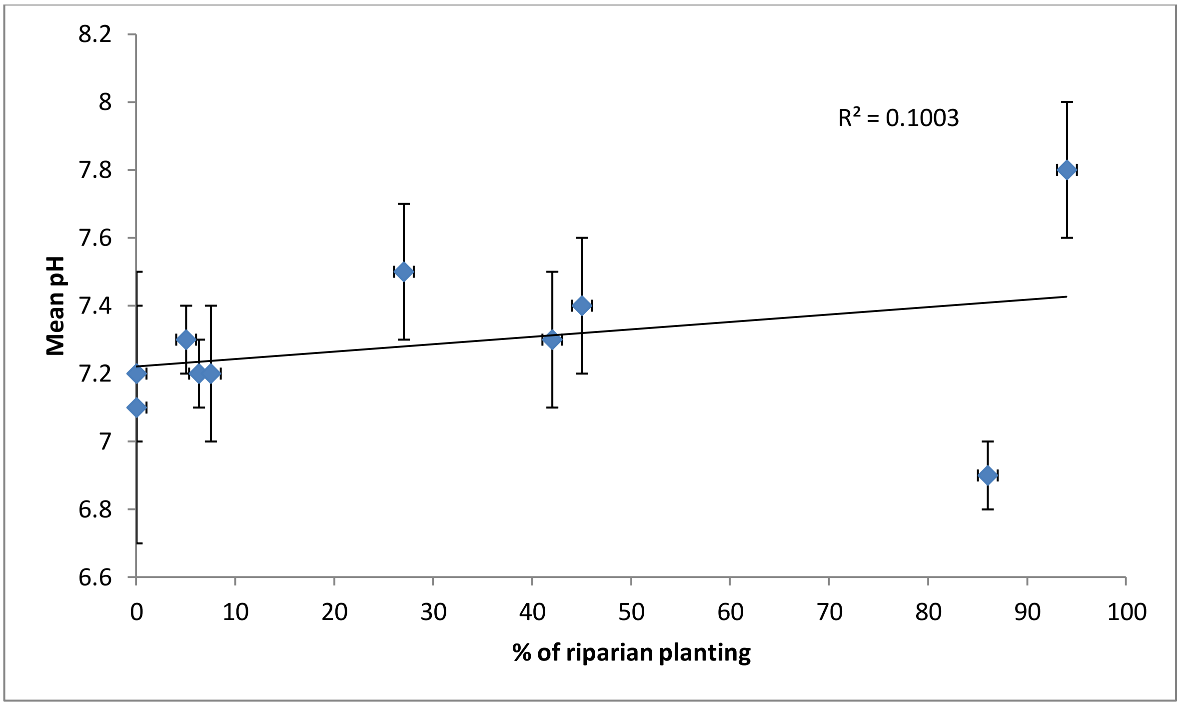

Fig. 3.3. The relationship between the mean pH (+/- S.E) and percentage of riparian planting for each site, across all sampling dates.

Seasonal variation effects

A single factor anova was carried out on all data to test for differences in pH across the seasons. There was a significant difference in pH between the seasons at α0.05 (F=7.7, d.f= 91, p= <0.001) (Figure 3.2). A least significance difference test was carried out to see which groups differed. At α0.05 significance, both summer and spring were significantly higher than autumn and winter. Summer and spring did not differ from each other and autumn and winter did not differ from each another.

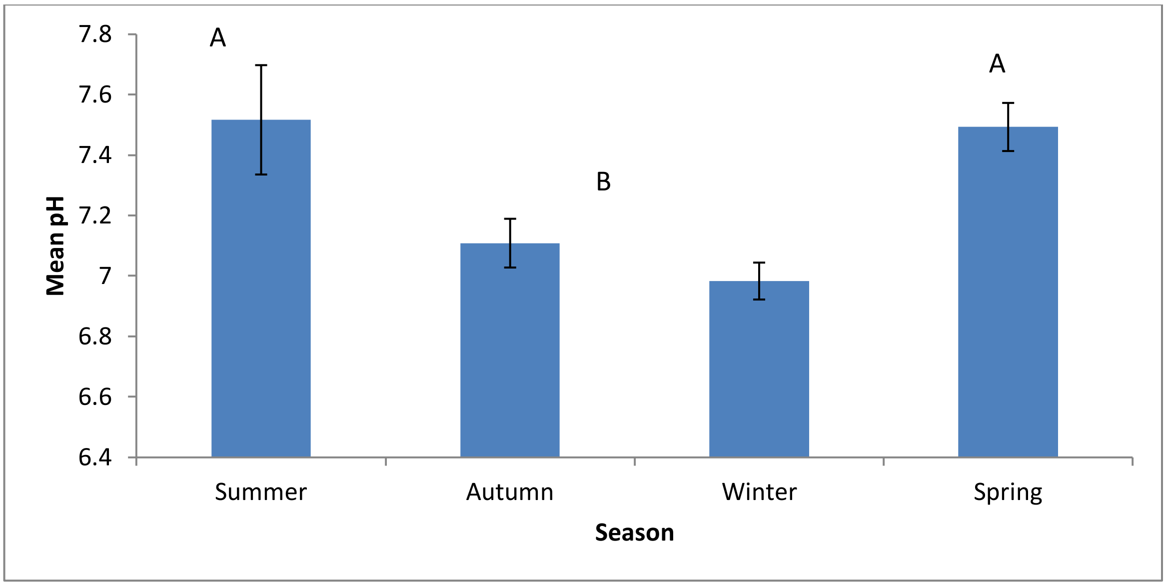

Fig. 3.4. Mean pH (+/- S.E) for each season, across all sampling dates. Letters indicate significantly different levels at α0.05.

3.2.3. Temperature

Across all sampling dates and all sites the lowest temperature recording was 5.5°C at site 5 and the highest was 20.3°C at site 8. No sites recorded extremely high temperatures throughout the duration of this study. The mean for each site ranged between 11.5°C at site 5 and 14.0°C at site 9 (Table 3.3).

Table 3.3. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for temperature at each sampling site over the sampling dates.

Site	Mean	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	13.1	10	8.7 - 19.3	4.1	1.3
2 (Embankment Rd.)	12.2	10	8.4 - 18.7	3.8	1.2
3 (Powells/Pannetts Rd.)	13.1	11	9.6 - 18.9	3.0	0.9
4 (Near Coes Ford)	12.2	8	8.8 - 16.8	2.9	1.0
5 (Wolf Creek)	11.5	8	5.5 - 17.9	4.5	1.6
6 (Hanmer Rd.)	12.2	11	7.5 - 18.3	3.7	1.1
7 (Colletts Rd.)	12.7	11	8.3 - 17.9	3.2	1.0
8 (Drain Rd.)	13.3	11	7.5 - 20.3	4.1	1.2
9 (Tramway Rd.)	14.0	11	8.9 - 19.6	3.3	1.0
10 (Off Lower Lake Rd.)	13.6	11	11.4 - 17.1	1.7	0.5

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between mean temperature and the percentage of riparian planting on each drain. There was no significant relationship between mean temperature and percentage of riparian planting along each site ($F=1.7$, $d.f=9$, $p= 0.2$) (Figure 3.5).

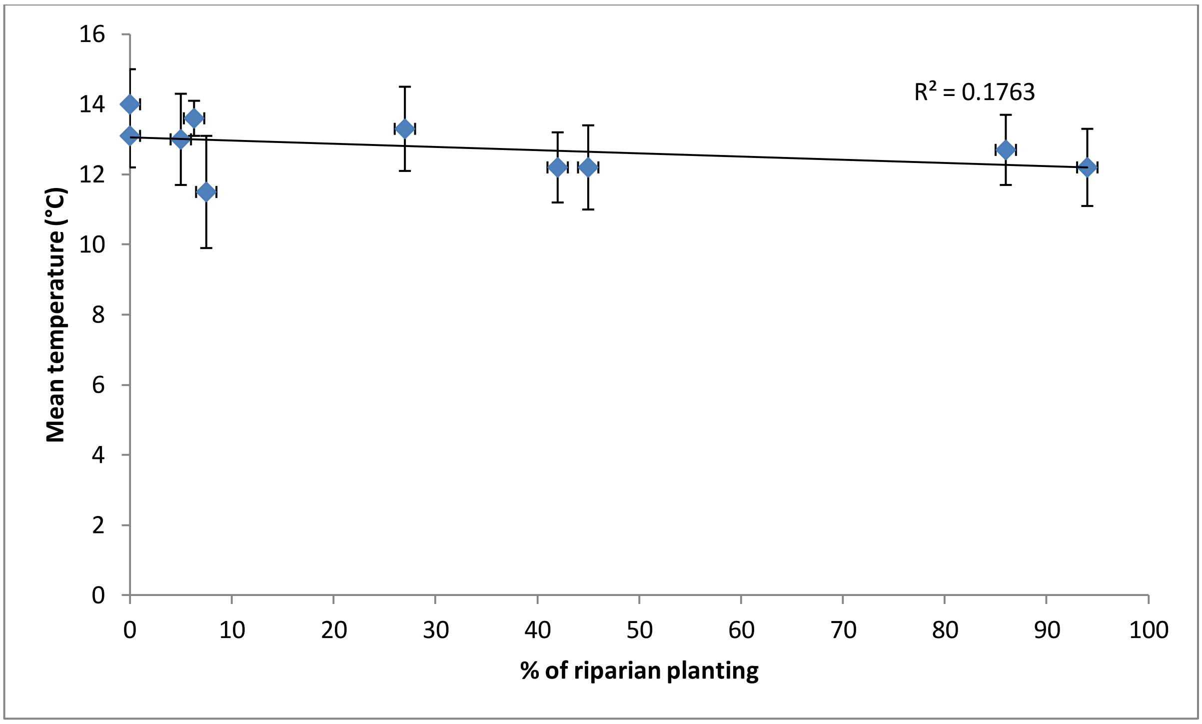

Fig. 3.5. Relationship between mean temperature (°C) (+/- S.E) and percentage of riparian planting for each site, across all sampling dates.

Season effects

A single factor anova was carried out on all data to test for differences in mean temperature across the seasons. There was a significant difference between temperatures at α0.05 (F=58.6, d.f= 101, p= <.001) (Figure 3.6). A least significance difference test was carried out to see which groups differed. At α0.05 significance, summer was significantly higher than all other seasons. Spring was significantly higher than autumn and winter. Autumn was significantly higher than winter.

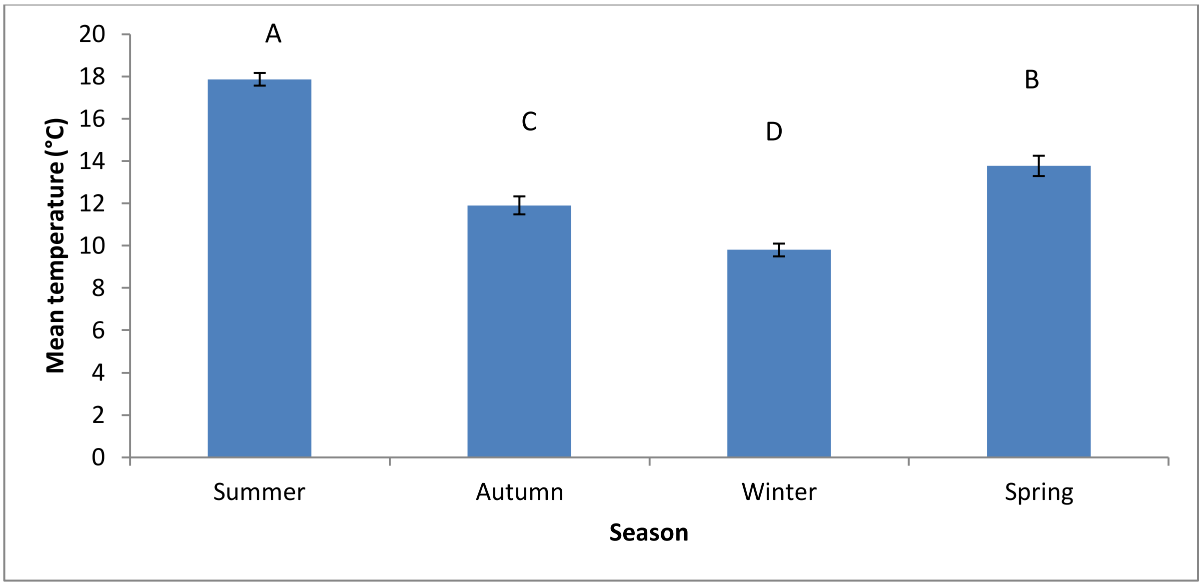

Fig. 3.6. Mean temperature (°C) (+/- S.E) for each season, across all sampling dates. Letters indicate significantly different levels at α0.05.

3.2.4. Conductivity

Across all sampling dates and all sites the lowest conductivity recording was 189 µS/cm at site 10 and the highest was 3500 µS/cm at site 1. The mean for each site ranged between 218 (µS/cm) at site 10 and 2256 (µS/cm) at site 1 (Table 3.4). Sites 1 and 2 are in the order of 10 times more ion-rich than the other drains.

Table 3.4. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for conductivity (µS/cm) at each sampling site over the sampling dates.

Site	Mean	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	2256	10	659 – 3500	754	238
2 (Embankment Rd.)	1969	10	330 – 2496	697	220
3 (Powells/Pannetts Rd.)	309	11	269 – 376	38	11
4 (Near Coes Ford)	358	8	293 - 525	80	28
5 (Wolf Creek)	390	8	319 - 501	55	19
6 (Hanmer Rd.)	378	11	282 - 510	97	29
7 (Colletts Rd.)	300	11	267 – 411	39	12
8 (Drain Rd.)	333	11	274 - 400	37	11
9 (Tramway Rd.)	312	11	273 - 375	30	9
10 (Off Lower Lake Rd.)	218	11	189 - 299	33	10

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between mean conductivity (µS/cm) and the percentage of riparian planting on each drain. There was no significant relationship between mean conductivity and percentage of riparian planting along each site ($F=0.08$, $d.f=9$, $p=0.8$) (Figure 3.7).

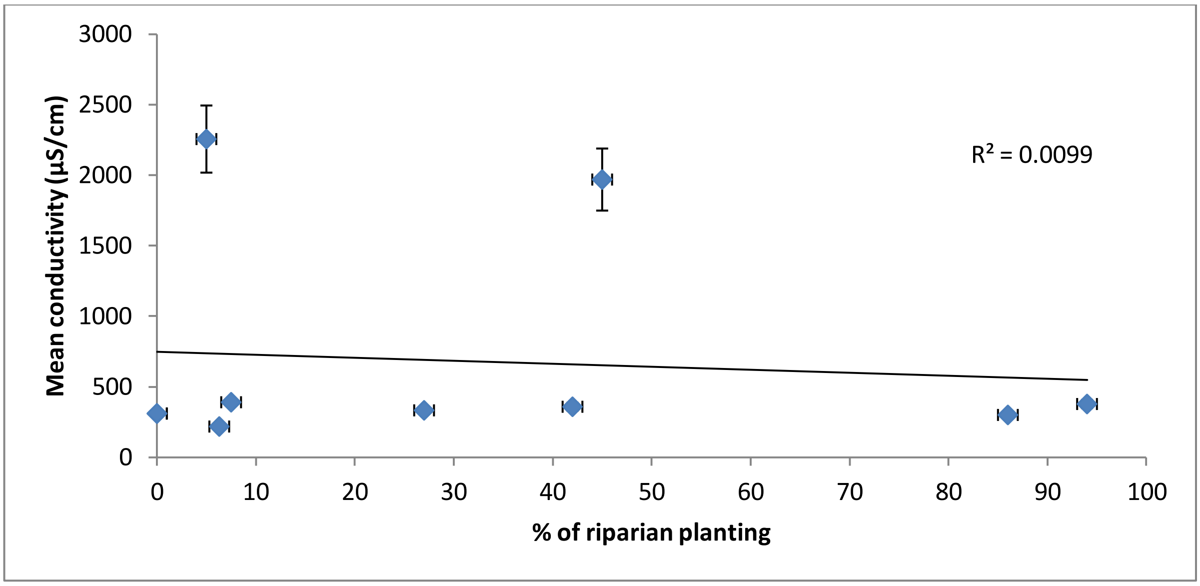

Fig. 3.7. Mean conductivity (µS/cm) (+/- S.E) and percentage of riparian planting for each site, across all sampling dates.

Season effects

A single factor anova was carried out on all data to test for differences in mean conductivity across the seasons. There was no significant difference between conductivity across the seasons at $\alpha 0.05$ ($F=0.1$, d.f= 101, $p=0.9$) (Figure 3.8).

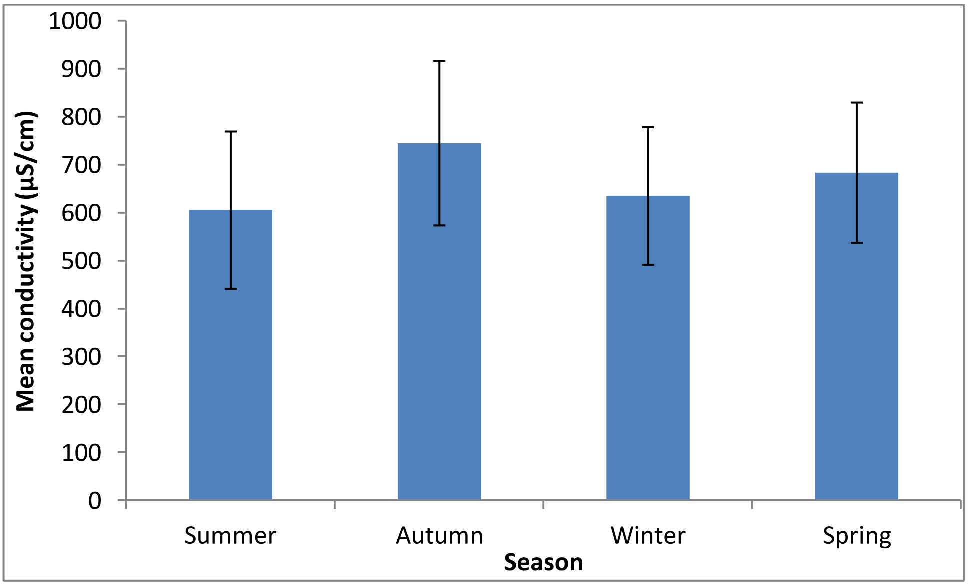

Fig.3.8. Mean conductivity (µS/cm) (+/- S.E) for each season, across all sampling dates.

3.3 Phosphorus

3.3.1. Total Phosphorus

Across all sampling dates and all sites the lowest TP concentration was 0.01 mg/L at sites 3, 7, 9 and 10 and the highest was 1.37 mg/L at site 1. The mean for each site ranged between 0.04 mg/L at sites 7 and 10 and 1.22 mg/L at site 1 (Table 3.5). Sites 1 and 2 had approximately 10 times greater TP concentrations than the other drains.

Table 3.5. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for TP concentrations (mg/L) at each sampling site over the sampling dates.

Site	Mean	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	1.22	6	0.93 - 1.37	0.18	0.07
2 (Embankment Rd.)	1.03	7	0.71 - 1.30	0.25	0.10
3 (Powells/Pannetts Rd.)	0.08	10	0.01 - 0.19	0.06	0.02
4 (Near Coes Ford)	0.11	8	0.04 - 0.18	0.05	0.02
5 (Wolf Creek)	0.13	8	0.09 - 0.22	0.05	0.02
6 (Hanmer Rd.)	0.08	9	0.03 - 0.20	0.05	0.02
7 (Colletts Rd.)	0.04	11	0.01 - 0.08	0.02	0.01
8 (Drain Rd.)	0.05	11	0.02 - 0.11	0.03	0.01
9 (Tramway Rd.)	0.07	11	0.01 - 0.15	0.06	0.02
10 (Off Lower Lake Rd.)	0.04	11	0.01 - 0.08	0.02	0.01

Figure 3.9 shows the TP concentrations (mg/L) at all sites over the entire sampling period. January is excluded from this graph as there were no drains that had sufficient water flow to justify sampling for TP at this time. A single factor anova was carried out on TP data to see if there were any differences between TP concentrations for each drain. There was a significant difference for $\alpha 0.05$ ($f=166.6$, $d.f=91$, $p= <0.001$). A LSD test was carried out to

see which drains were different. At α0.05 sites 1 and 2 had significantly higher TP concentrations than all the other sites. All the other sites did not significantly differ from one another (Figure 3.10).

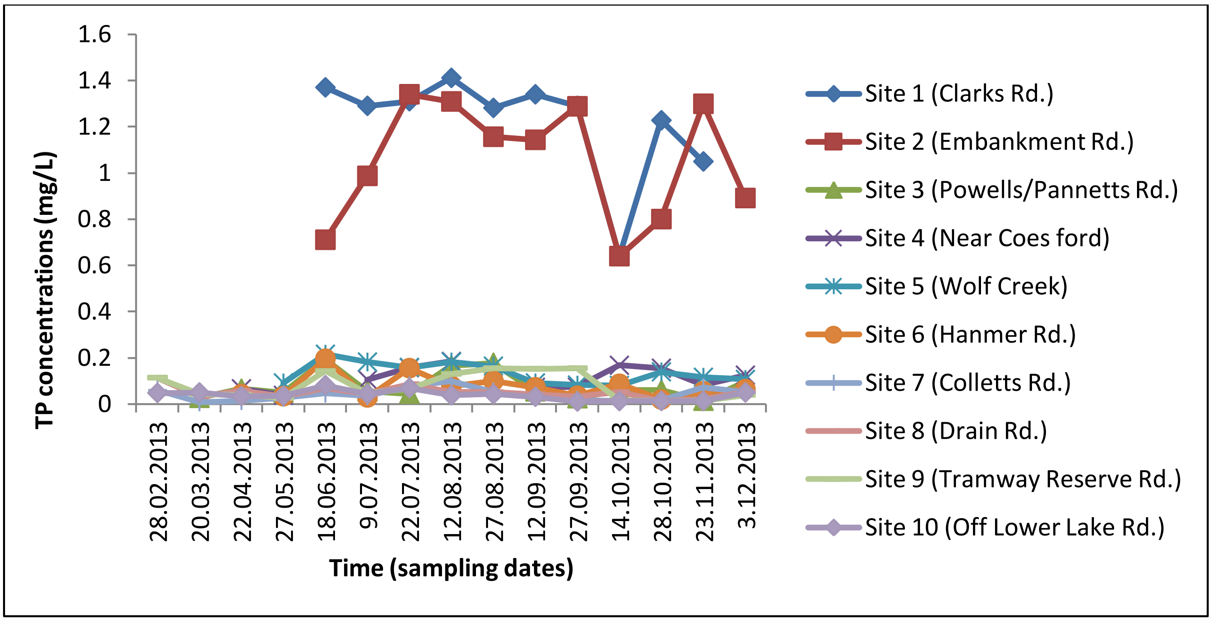

Fig. 3.9. TP concentrations (mg/L) for each site over the entire sampling period.

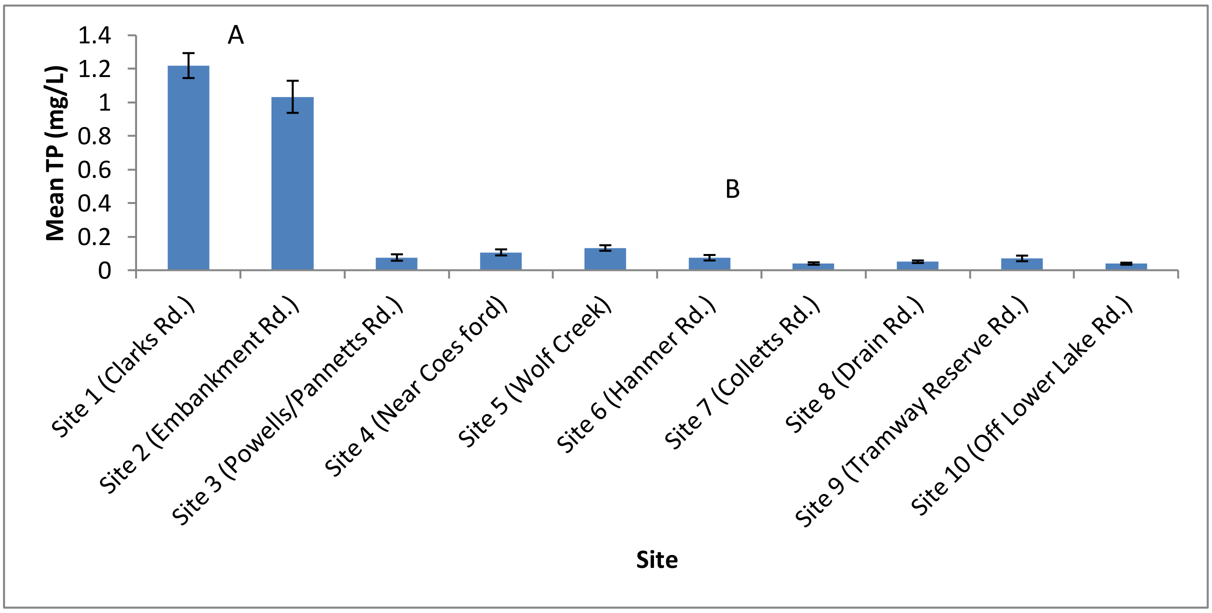

Fig. 3.10. Mean TP concentrations (mg/L) (+/- S.E) for each site, over the entire sampling period. Letters indicate significantly different concentrations at α0.05.

3.3.2. Riparian planting vs effects

A linear model regression was conducted to test for a relationship between average TP concentrations (mg/L) and the percentage of riparian planting on each drain. There was no significant relationship between average TP and percentage of riparian planting along each site at α0.05 (F= 0.1, d.f=9, p= 0.7) (Figure 3.11).

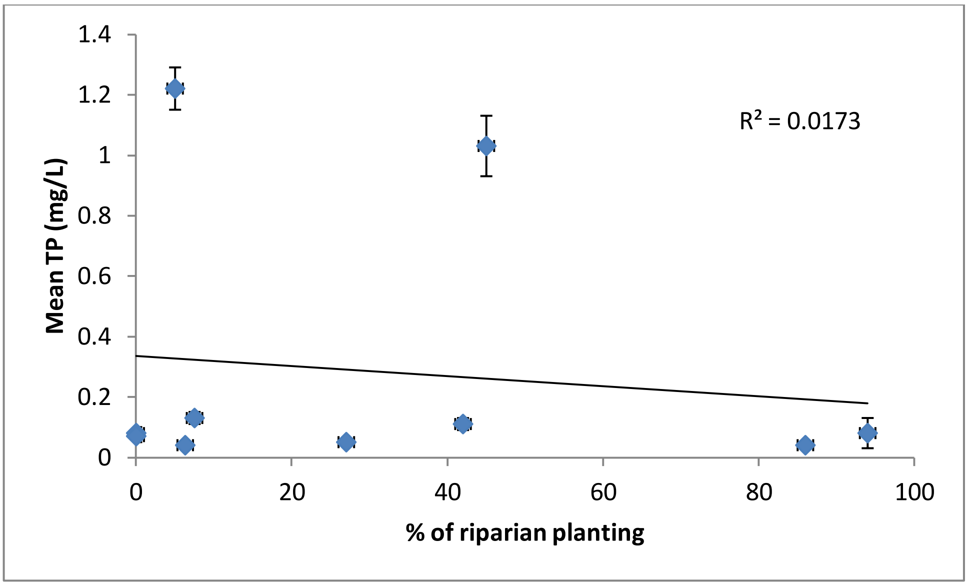

Fig. 3.11. The relationship between the mean TP concentrations (mg/L) (+/- S.E) and the percentage of riparian planting for each site, across sampling dates

3.3.2. Dissolved reactive phosphorus (DRP)

Table 3.6 shows the percentages of TP that were present as the form of DRP (i.e, in dissolved phosphate, PO_4) (NB: DRP data are shown in appendix only). Only data that had higher TP values than 0.1 mg/L were included (site 10 had no TP values higher than 0.1 mg/L so it is not included in the table). The lowest mean DRP was 52% of TP at site 6, and the highest mean DRP was 96% at site 9. The overall lowest data DRP mean was 29% at site 6 and the overall highest DRP mean was 100% at sites 2, 3, 4, 8 and 9. The mean DRP percentage of TP over all drains was 69%.

Table 3.6. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for DRP percentages for each TP values of more than 0.1mg/L over the sampling dates.

Site	Mean	N	Min. - Max. (%)	S.D	S.E
1 (Clarks Rd.)	61	10	44 - 96	17.6	5.6
2 (Embankment Rd.)	62	11	38 - 100	27.4	8.3
3 (Powells/Pannetts Rd.)	69	4	39 - 100	24.9	12.4
4 (Near Coes Ford)	76	6	49 - 100	17.4	7.1
5 (Wolf Creek)	71	7	43 - 96	18.7	7.1
6 (Hanmer Rd.)	52	3	29 - 77	23.6	13.6
7 (Colletts Rd.)	60	1	60	-	-
8 (Drain Rd.)	96	3	94 - 100	3.4	2.0
9 (Tramway Rd.)	72	3	52 - 100	25.8	14.9

3.4 Suspended particulate matter (SPM)

Across all sampling dates and all sites the lowest SPM concentration was 0.006 g/L at sites 3, 7 and 10 and the highest was 0.057 g/L at site 2. The mean for each site ranged between 0.010 g/L at site 10 and 0.044 g/L at site 2 (Table 3.7).

Table 3.7. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for SPM concentrations (g/L) at each sampling site over the entire sampling period.

Site	**Mean**	**N**	**Min. - Max.**	**S.D.**	**S.E.**
1 (Clarks Rd.)	0.033	6	0.016 - 0.050	0.013	0.005
2 (Embankment Rd.)	0.044	7	0.023 - 0.057	0.013	0.005
3 (Powells/Pannetts Rd.)	0.014	10	0.006 - 0.025	0.006	0.002
4 (Near Coes Ford)	0.019	8	0.007 - 0.046	0.013	0.005
5 (Wolf Creek)	0.029	8	0.012 - 0.039	0.009	0.003
6 (Hanmer Rd.)	0.018	9	0.009 - 0.031	0.010	0.003
7 (Colletts Rd.)	0.011	11	0.006 - 0.030	0.007	0.002
8 (Drain Rd.)	0.011	11	0.008 - 0.018	0.004	0.001
9 (Tramway Rd.)	0.011	11	0.008 - 0.021	0.004	0.001
10 (Off Lower Lake Rd.)	0.010	11	0.006 - 0.008	0.004	0.001

Figure 3.12 shows the SPM concentrations (g/L) at all sites over the entire sampling period. January is excluded from this graph as there were no drains that had any or sufficient water fow to justify sampling for SPM at this time. A single factor anova was carried out on SPM data to see if there were any differences between SPM concentrations for each drain. There was a significant difference for α0.05 (f=15.1, d.f=91, p= <0.001). A LSD test was carried out to see which drains were different. At α0.05 sites 2 had significantly higher SPM concentrations than all other sites. Sites 1 and 5 had significantly higher SPM concentrations than all other sites (except site 2). All the other sites did not differ from one another (Figure 3.13).

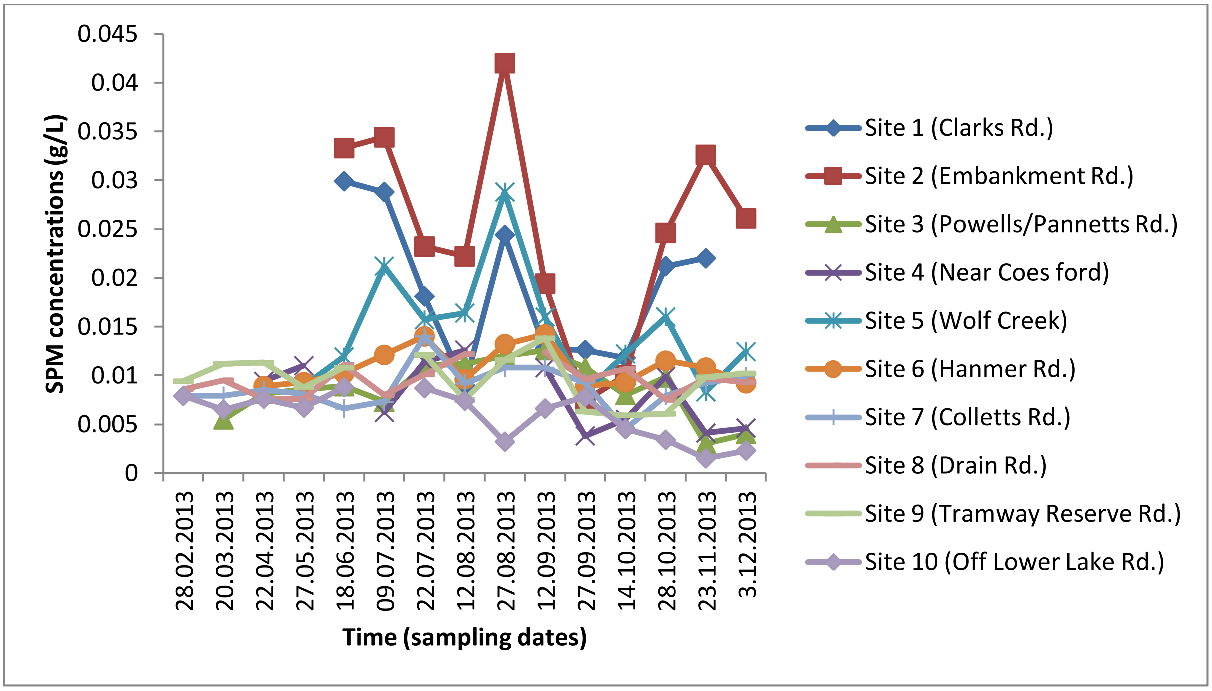

Fig 3.12.SPM concentrations (g/L) for each site, across the entire sampling period.

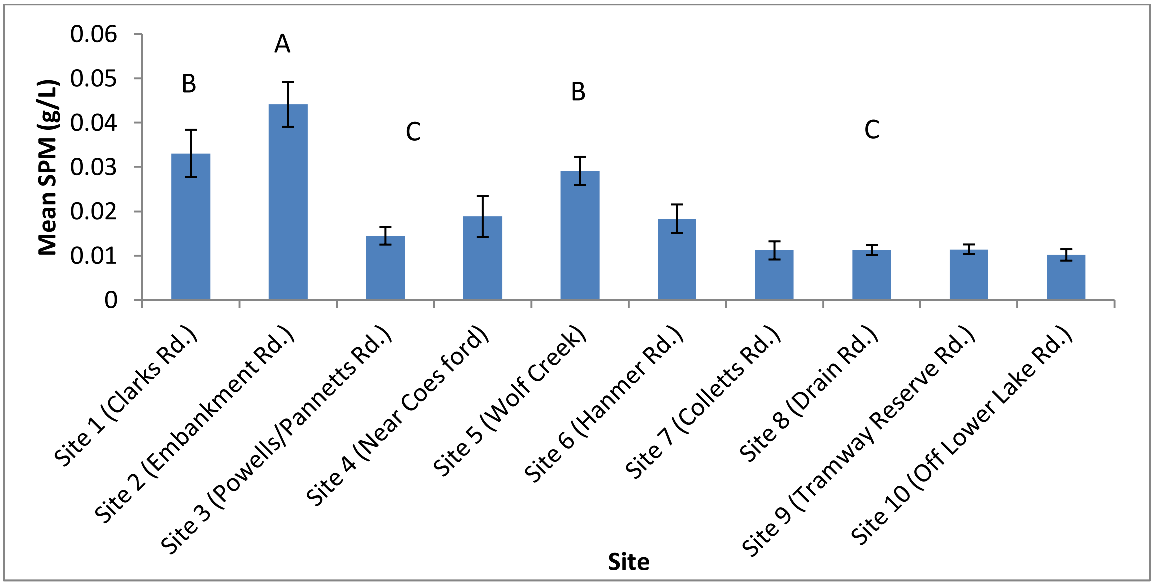

Fig. 3.13. Mean SPM concentrations (g/L) (+/- S.E) across all sites. Letters indicate significant differences at α0.05.

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between average SPM concentrations (g/L) and the percentage of riparian planting on each drain. There was no significant relationship between mean SPM and percentage of riparian planting along each site at α0.05 (F=.006, d.f=9, p= 0.9) (Figure 3.14).

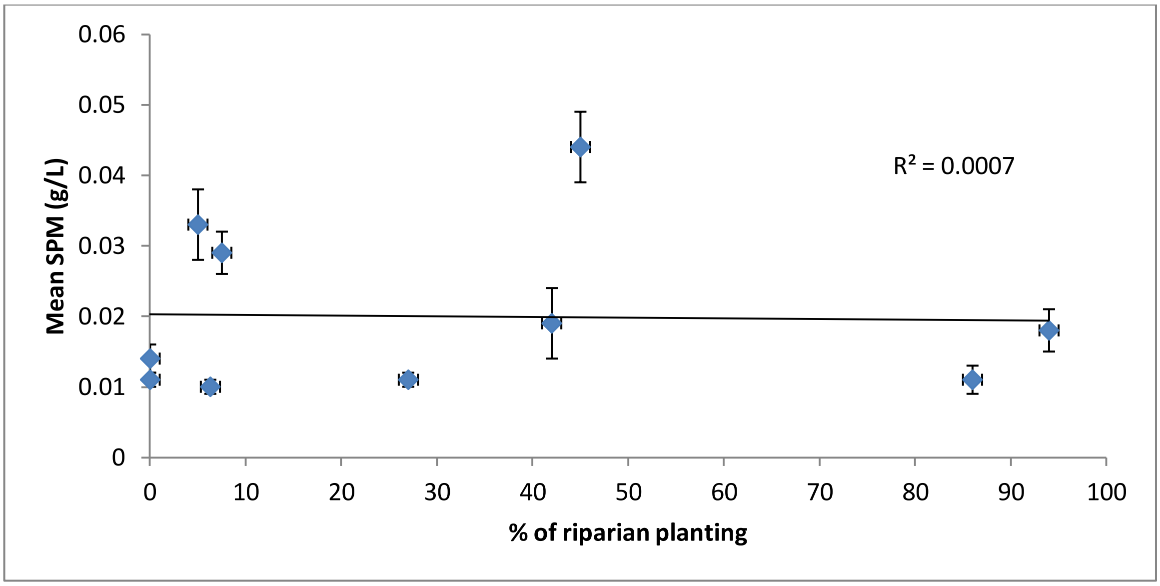

Fig. 3.14. Mean SPM concentrations (g/L) (+/- S.E) and the percentage of riparian planting over all sampling dates.

3.5 Macroinvertebrate results

3.5.1. Species Richness

Across all sampling dates and all sites the lowest macroinvertebrate species richness was 2 at site 1 and the highest was 13 at site 6. The averages for each site ranged between 4.8 at site 1 and 8.4 at site 6 (Table 3.8).

Table 3.8. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E) for macroinvertebrate species richness at each sampling site over the entire sampling period.

Site	Mean	N	Min. - Max.	S.D	S.E
1 (Clarks Rd.)	4.8	10	2 - 7	1.75	0.55
2 (Embankment Rd.)	5.0	9	3 - 7	1.50	0.50
3 (Powells/Pannetts Rd.)	7.5	10	5 - 11	2.22	0.70
4 (Near Coes Ford)	7.7	6	4 - 10	2.58	1.05
5 (Wolf Creek)	5.0	8	4 - 7	1.31	0.44
6 (Hanmer Rd.)	8.4	10	6 - 13	2.55	0.81
7 (Colletts Rd.)	5.7	10	4 - 7	0.82	0.26
8 (Drain Rd.)	8.2	9	7 - 9	0.83	0.28
9 (Tramway Rd.)	8.0	10	7 - 10	0.94	0.30
10 (Off Lower Lake Rd.)	5.8	10	4 - 7	1.14	0.36

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between average species richness and the percentage of riparian planting on each drain. There was no significant relationship between average species richness and percentage of riparian planting along each site at $\alpha 0.05$ ($F= 0.23$, d.f=9, $p= 0.63$) (Figure 3.15).

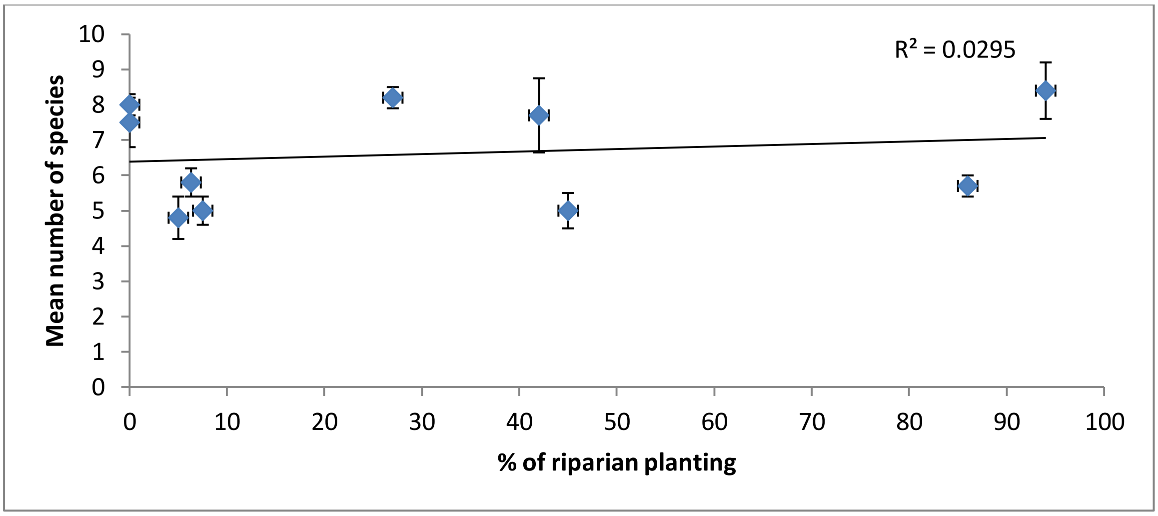

Fig. 3.15. Mean macroinvertebrate species richness and percentage of riparian planting across all sites, over the entire sampling period (+/- S.E).

Substrate effects

A single factor anova was carried out on all data to test for differences in species richness across the different substrates found in the drains. There was a significant difference at α0.05 (F=17.67, d.f= 91, p= <0.001) (Figure 3.16). A least significance difference test was carried out to see which groups differed. At α0.05 significance, silt/gravel (S/G) and cobble (C) had significantly higher species richness than silt/sand/mud/impervious (S/S/M/I). S/G and C did not differ.

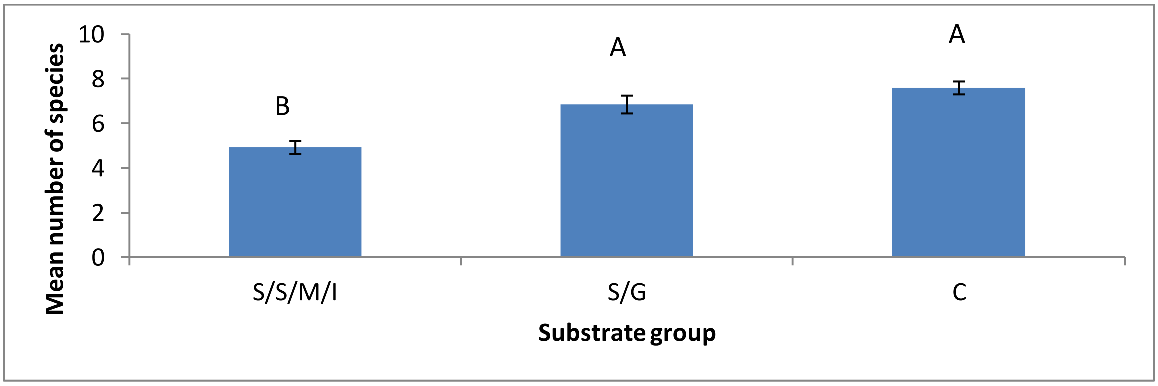

Fig. 3.16. Mean species richness for each substrate group (+/- S.E). S/S/M/I: silt/sand/mud/impervious, S/G: silt/gravel, C: cobbles. Letters indicate significantly different groups at α0.05.

3.5.2. EPT%

Across all sampling dates and all sites the lowest EPT% was 0% at sites 1, 2, 5 and 7 and the highest was 55.6% at site 9. The averages for each site ranged between 0% at site 1 and 38.1% at site 4 (Table 3.9).

Table 3.9. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E) for EPT% at each sampling site over the entire sampling period.

Site	Mean (%)	N	Min. - Max.	S.D	S.E
1 (Clarks Rd.)	0.0	10	0 - 0	0	0
2 (Embankment Rd.)	10.2	9	0 - 16.7	10.0	3.34
3 (Powells/Pannetts Rd.)	28.2	10	14.3 - 37.5	8.41	2.66
4 (Near Coes Ford)	38.1	6	25 - 50	8.33	3.40
5 (Wolf Creek)	18.4	8	0 - 28.6	11.82	3.94
6 (Hanmer Rd.)	32.2	10	16.67 - 40	7.97	2.52
7 (Colletts Rd.)	30.8	10	0 - 40	11.34	3.59
8 (Drain Rd.)	30.9	9	25 - 37.5	4.29	1.43
9 (Tramway Rd.)	34.8	10	25 - 55.6	9.50	3.00
10 (Off Lower Lake Rd.)	37.3	10	16.7 - 42.86	8.99	2.84

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between average EPT% and the percentage of riparian planting on each drain. There was no significant relationship between average EPT% and percentage of riparian planting along each site at α0.05 ($F= 0.33$, $d.f=9$, $p= 0.58$) (Figure 3.17).

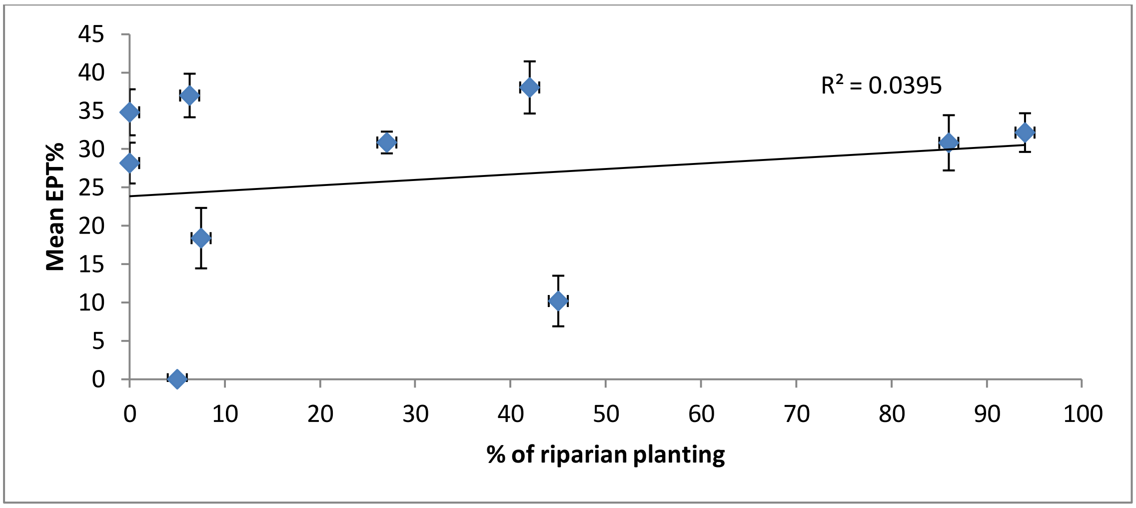

Fig. 3.17. Mean EPT% against percentage of riparian planting across all sites over the entire sampling period (+/- S.E).

Substrate effects

A single factor anova was carried out on all data to test for differences in EPT% across the different substrates found in the drains. There was a significant difference at α0.05 ($F=59.92$, d.f= 91, $p= <0.001$) (Figure 3.18). A least significance difference test was carried out to see which groups differed. At α0.05 significance, S/G and C substrates had significantly higher EPT% than S/S/M/I substrate. S/G and C substrate did not differ.

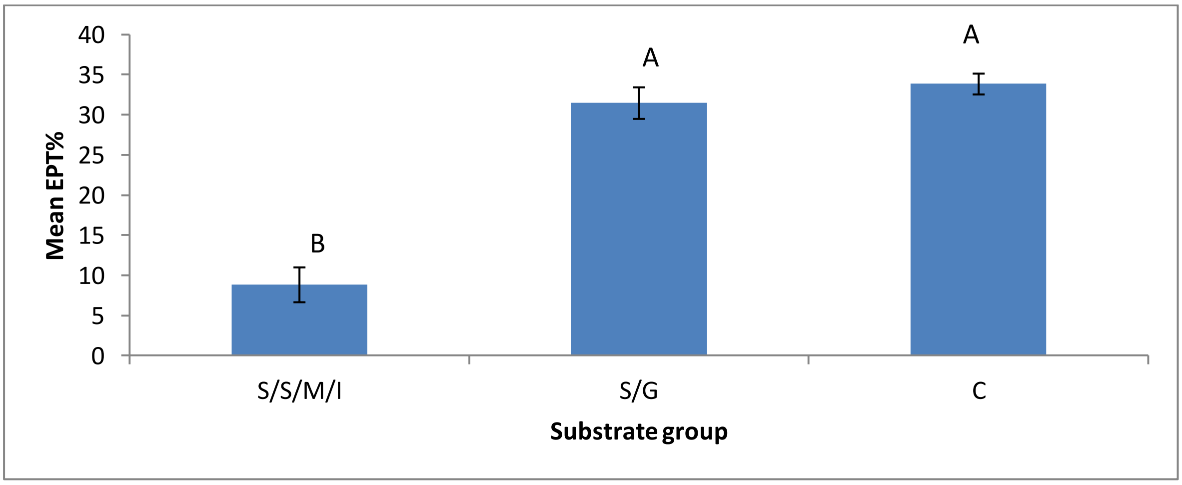

Fig. 3.18. Mean EPT% for each substrate group (+/- S.E). S/S/M/I: silt/sand/mud/impervious, S/G: silt/gravel, C: cobbles. Letters indicate significantly different groups at α0.05.

3.5.3. Macroinvertebrate community index (MCI)

Across all sampling dates and all sites the lowest MCI score was 40 at sites 1 and 5 and the highest was 104 at site 4. The mean for each site ranged between 52 at site 1 and 84 at site 4 (Table 3.10).

Table 3.10. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E) for MCI scores at each sampling site over the entire sampling period.

Site	Mean	N	Min. - Max.	S.D	S.E
1 (Clarks Rd.)	52	10	40 - 65	8.27	2.62
2 (Embankment Rd.)	54	9	45 - 67	5.78	1.93
3 (Powells/Pannetts Rd.)	71	10	48 - 87	10.72	3.39
4 (Near Coes Ford)	84	6	60 - 104	20.26	8.27
5 (Wolf Creek)	53	8	40 - 80	14.00	4.67
6 (Hanmer Rd.)	77	10	68 - 90	6.94	2.19
7 (Colletts Rd.)	72	10	63 - 91	10.01	3.17
8 (Drain Rd.)	71	9	54 - 84	9.50	3.17
9 (Tramway Rd.)	77	10	63 - 100	11.32	3.58
10 (Off Lower Lake Rd.)	84	10	56.7 - 100	12.45	3.94

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between mean MCI scores and the percentage of riparian planting on each drain. There was no significant relationship between average MCI score and percentage of riparian planting along each site at $\alpha 0.05$ ($F= 0.30$, $d.f=9$, $p= 0.60$) (Figure 3.19).

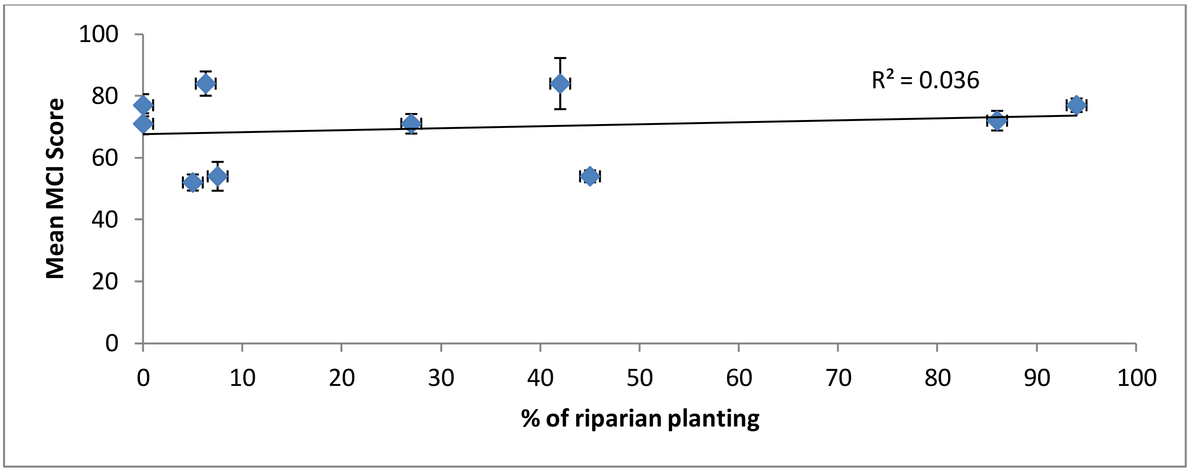

Fig. 3.19. Mean MCI score and percentage of riparian planting across all sites over the entire sampling period (+/- S.E).

Substrate effects

A single factor anova was carried out on all data to test for differences in MCI scores across the different substrates found in the drains. There was a significant difference at α0.05 (F=39.05, d.f= 91, p= <0.001) (Figure 3.20). A least significance difference test was carried out to see which groups differed. At α0.05 significance S/G and C substrates had significantly higher MCI scores than S/S/M/I substrate. S/G and C substrate did not differ.

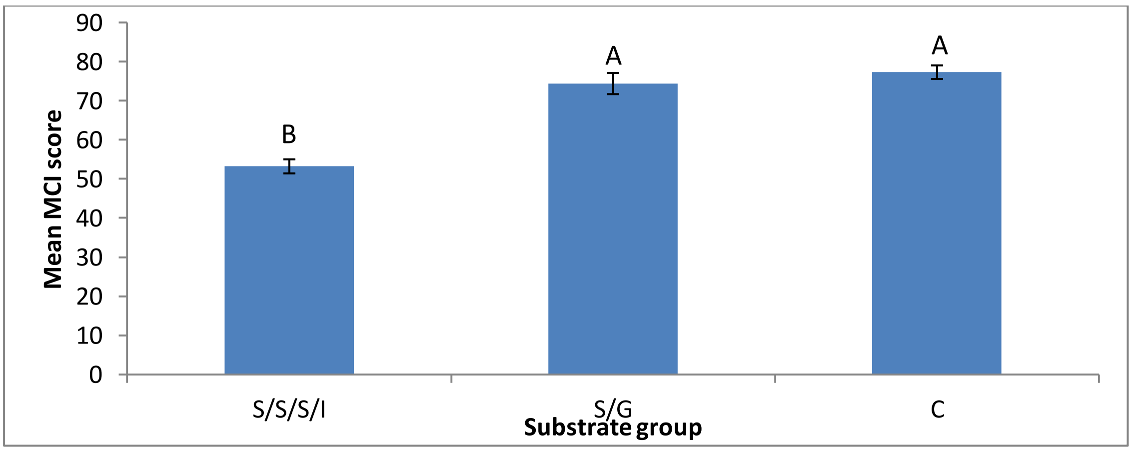

Fig. 3.20. Mean MCI score for each substrate group (+/- S.E). S/S/M/I: silt/sand/mud/impervious, S/G: silt/gravel, C: cobbles. Letters indicate significantly different groups at α0.05.

3.5.4. Semi-quantative macroinvertebrate community index (SQMCI)

Across all sampling dates and all sites the lowest SQMCI score was 1.1 at site 8 and the highest was 6.44 at site 4. The mean for each site ranged between 2.11 at site 5 and 3.98 at site 4 (Table 3.11).

Table 3.11. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E) for SQMCI scores at each sampling site over the entire sampling period.

Site	Mean	N	Min. - Max.	S.D	S.E
1 (Clarks Rd.)	2.52	10	1.50 - 2.83	0.53	0.17
2 (Embankment Rd.)	2.76	9	1.97 - 3.25	0.47	0.16
3 (Powells/Pannetts Rd.)	2.95	10	1.72 - 3.80	0.72	0.23
4 (Near Coes Ford)	3.98	6	2.54 - 6.44	1.35	0.55
5 (Wolf Creek)	2.11	8	1.51 - 3.53	0.62	0.21
6 (Hanmer Rd.)	3.12	10	2.26 - 3.95	0.60	0.19
7 (Colletts Rd.)	3.13	10	2.00 - 3.98	0.68	0.22
8 (Drain Rd.)	2.94	9	1.10 - 3.81	1.12	0.37
9 (Tramway Rd.)	3.23	10	1.42 - 4.37	0.98	0.31
10 (Off Lower Lake Rd.)	3.45	10	1.74 - 4.97	1.13	0.36

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between average SQMCI scores and the percentage of riparian planting on each drain. There was no significant relationship between average SQMCI scores and percentage of riparian planting along each site at α0.05 ($F= 0.45$, $d.f=9$, $p= 0.52$) (Figure 3.21).

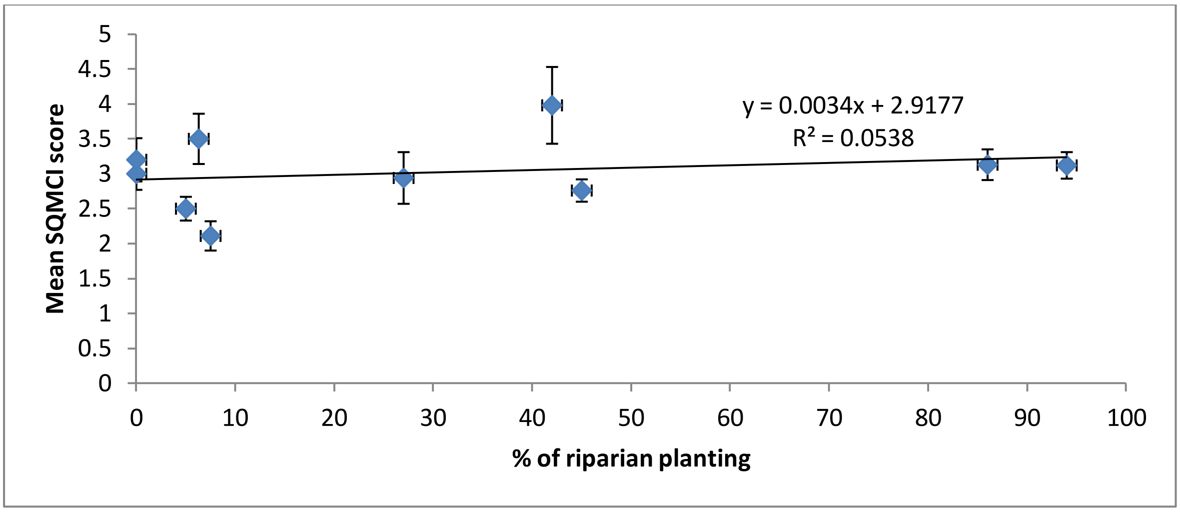

Fig. 3.21. Mean SQMCI score and percentage of riparian planting across all sites over the entire sampling period (+/- S.E).

Substrate effects

A single factor anova was carried out on all data to test for differences in SQMCI scores across the different substrates found in the drains. There was a significant difference at α0.05 (F=7.13, d.f= 91, p= 0.001) (Figure 3.22). A least significance difference test was carried out to see which groups differed. At α0.05 significance, SG and C substrates had significantly higher SQMCI scores than S/S/M/I substrate. S/G and C substrate did not differ.

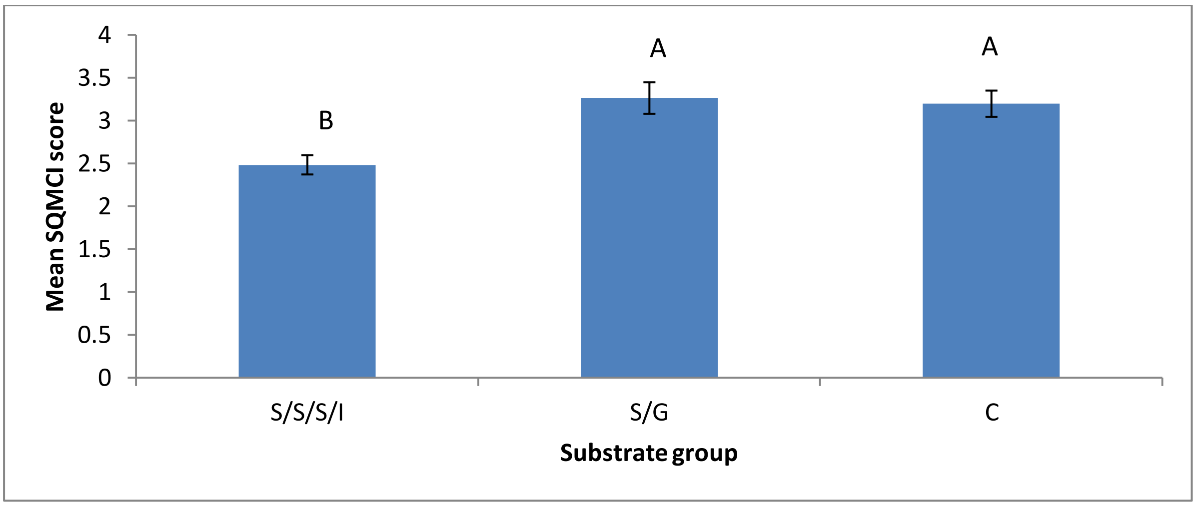

Fig. 3.22. Mean SQMCI score for each substrate group (+/- S.E). S/S/M/I: silt/sand/mud/impervious, S/G: silt/gravel, C: cobbles. Letters indicate significantly different groups at α0.05.

3.6 Flow

Across all sampling dates and sites the lowest flow recordings were 0.00 (L/s) at all sites (most often in January) and the highest was 1764.3 (L/s) at site 6. The averages for each site ranged between 6.8 (L/s) at site 1 and 411 (L/s) at site 6 (Table 3.12).

Table 3.12. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for flow recordings (L/s) at each sampling site excluding high flow events.

Site	Mean (L/s)	N	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	6.8	11	0.0 – 72.6	20.8	6.0
2 (Embankment Rd.)	13.1	11	0.0 – 143.8	41.3	11.9
3 (Powells/Pannetts Rd.)	210.1	11	0.0 – 557.8	177.0	51.1
4 (Near Coes Ford)	255.0	10	0.0 – 798.0	322.5	97.2
5 (Wolf Creek)	70.7	11	0.0 – 314.7	105.5	30.4
6 (Hanmer Rd.)	411.0	11	0.0 – 1764.3	520.4	150.2
7 (Colletts Rd.)	9.7	11	0.0 – 24.2	7.9	2.3
8 (Drain Rd.)	279.0	11	0.0 – 1052.4	313.0	90.4
9 (Tramway Rd.)	248.3	11	0.0 – 1203.0	330.2	95.3
10 (Off Lower Lake Rd.)	59.9	11	0.0 – 205.1	58.0	16.7

Figure 3.23 shows the flow (L/s) for each drain over the year and all sites, clearly identifying the high flow events that occurred in winter.

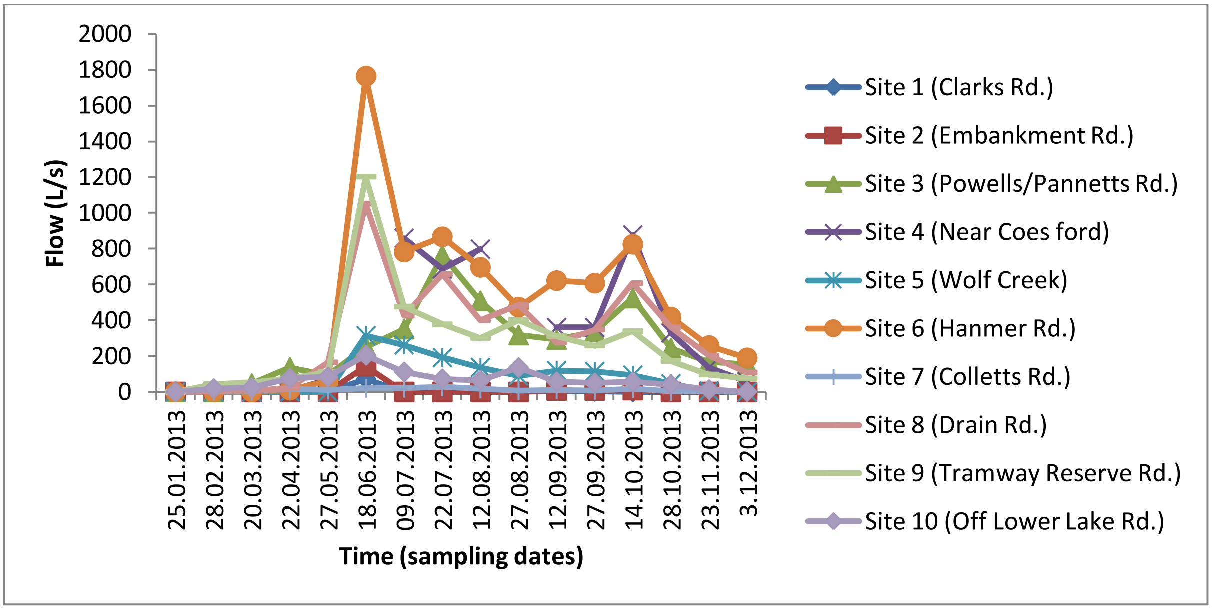

Fig 3.23. Flow levels (L/s) for each drain over time.

A single factor anova was carried out on flow data to see if there were any differences in flow between drains. There was a significant difference for α0.05 (f=3.9, d.f=118, p= <0.001). A LSD test was carried out to see which drains were different. At α0.05 sites 3, 4, 6, 8 and 9 had significantly higher flow than all other drains. Sites 5 and 10 had higher flows than sites 1, 2 and 7. Sites 1, 2 and 7 did not differ (fig 3.24).

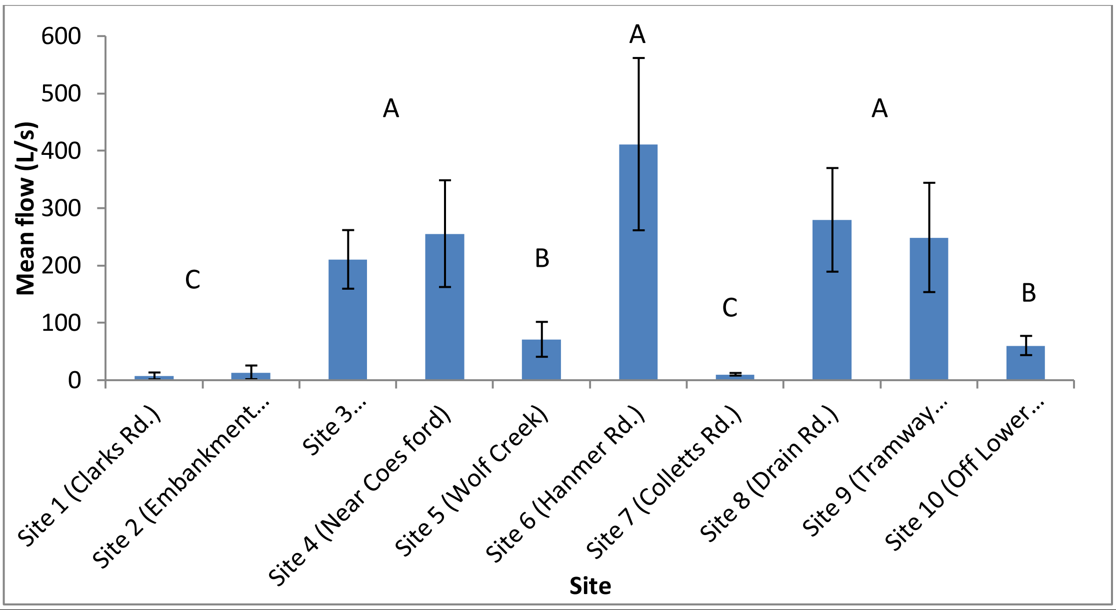

Fig. 3.24. Mean flow (L/s) (+/- S.E) for each drain. Letters indicate significant differences at α0.05.

Seasonal effects

A single factor anova was carried out on all data to test for differences in flow (L/s) levels across seasons. There was a significant difference for flow levels between the seasons at α0.05 (F=15.5, d.f= 118, p= <0.001) (Figure 3.25). A least significance difference test was carried out to see which seasons significantly differed. At α0.05 significance, winter was significantly higher than all other seasons. Spring was significantly higher than autumn and summer. Summer and autumn did not differ from each other.

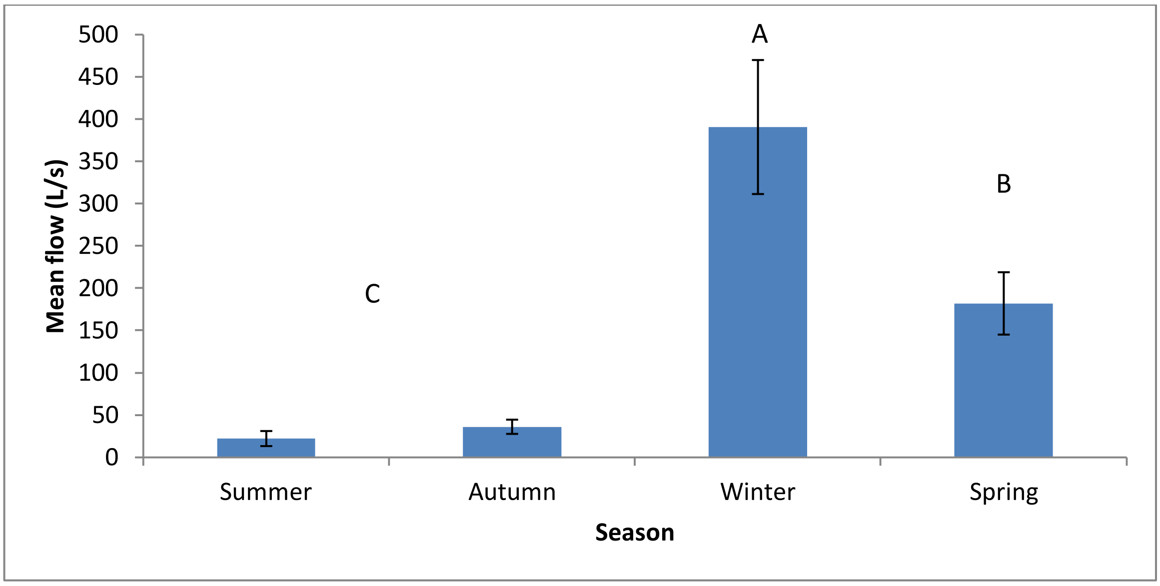

Fig. 3.25. Mean flow (L/s) (+/- S.E) for each season across the year. The letters indicate significant differences at α0.05.

3.7 Load

3.7.1. Total Phosphorus

Load values were calculated excluding June data. The mean TP load for each site ranged between 11 kg/year at site 7 and 1114 kg/year at site 4 (Table 3.26). The mean annual load of TP across all the drains was 326 kg/year per drain. Using the mean DRP percentage value of 69% of TP, approximately 225 kg/year is therefore in dissolved form.

Table 3.26. Mean, number of samples (N), min. - max. values, standard deviation (S.D) and standard error (S.E) for TP loads (kg/year) at each sampling site..

Site	Mean	N	Min. - Max.	S.D	S.E
1 (Clarks Rd.)	36	11	0 - 178	80	24
2 (Embankment Rd.)	40	11	0 - 310	977	29
3 (Powells/Pannetts Rd.)	473	11	0 - 2220	650	196
4 (Near Coes Ford)	1114	10	0 - 4581	1653	499
5 (Wolf Creek)	217	11	0 - 1205	383	115
6 (Hanmer Rd.)	631	11	0 - 2354	792	239
7 (Colletts Rd.)	11	11	0 - 43	13.2	4
8 (Drain Rd.)	292	11	0 - 1083	105	105
9 (Tramway Rd.)	395	11	0 - 1563	172	172
10 (Off Lower Lake Rd.)	54	11	0 - 162	54	16

Figure 3.26 shows the TP loads (kg/month) for each drain, over the year, identifying the increase over the winter months.

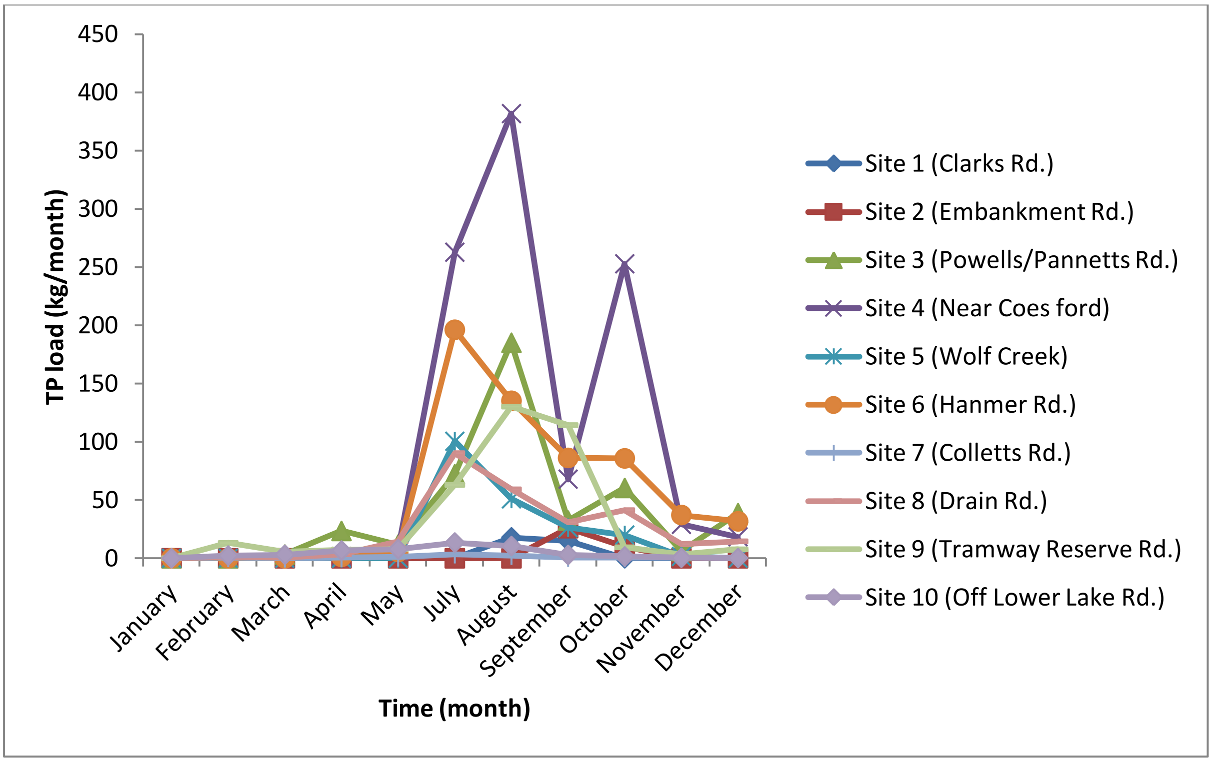

Fig. 3.26. TP loads (kg/month) over time (month). Note: Does not include June data.

A single factor anova was carried out on flow data (excluding June) to see if there were any differences in loads between drains. There was a significant difference for α0.05 (F= 3.0, d.f= 109, p = 0.003). A LSD test was carried out to see which drains were different. At α0.05 sites 4 and 6 had the highest loads and did not differ. Sites 6, 3, 5, 8 and 9 did not differ from one another but only site 6 was significantly higher than sites 1, 2, 7 and 10. Sites 1, 2, 7 and 10 did not differ from one another (Figure 3.27).

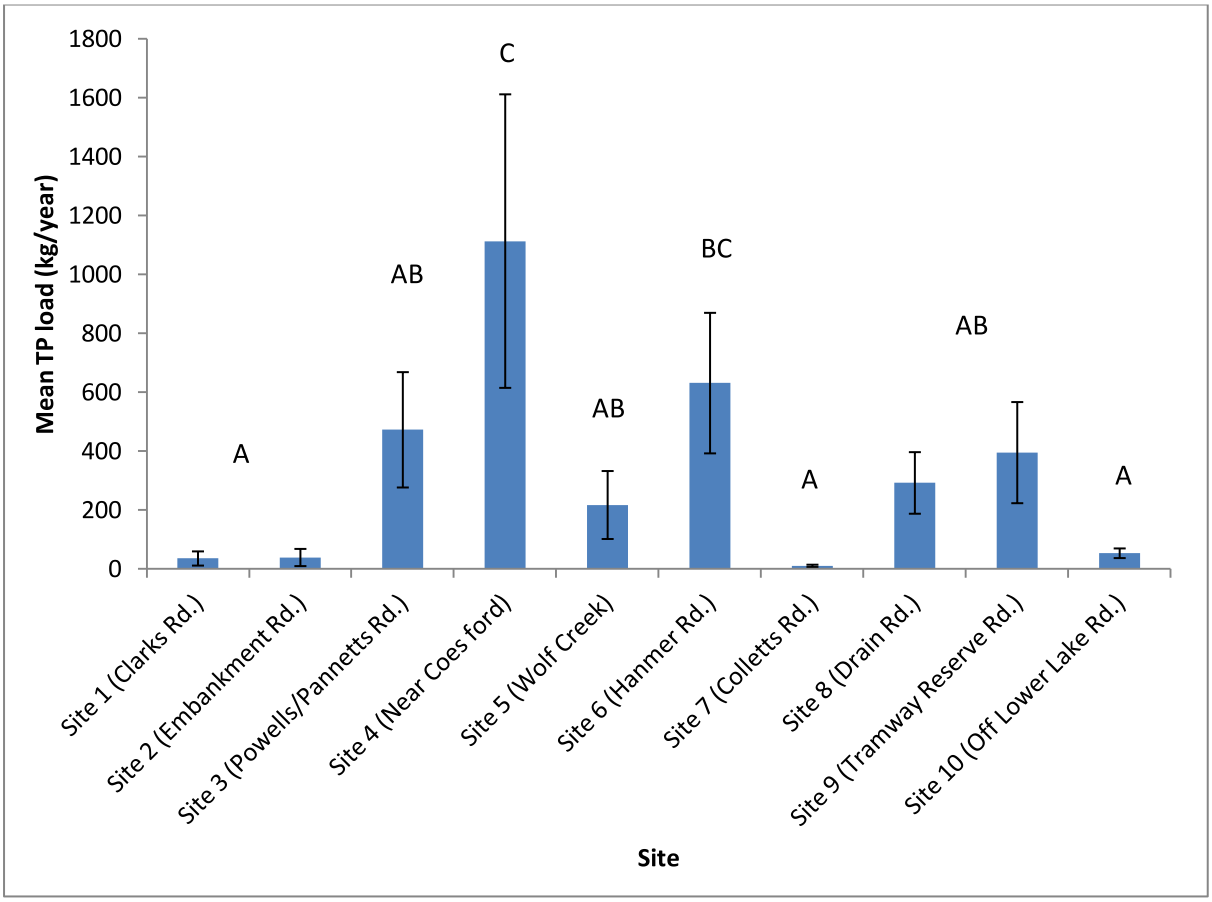

Fig. 3.27. Mean TP load (kg/year) (+/- S.E) for every site. Note: does not include June data. Letters indicate significant differences at α0.05.

Seasonal effects

A single factor anova was carried out on data (excluding June) to test for differences in TP loads (kg/day) across seasons. There was a significant difference for flow levels between the seasons at α0.05 ($F=13.8$, d.f = 109, $p=<0.001$) (Figure 3.28). A least significance difference test was carried out to see which seasons significantly differed. At α0.05 significance, winter was significantly higher than all other seasons. Spring was significantly higher than autumn and summer. Summer and autumn did not differ from each other.

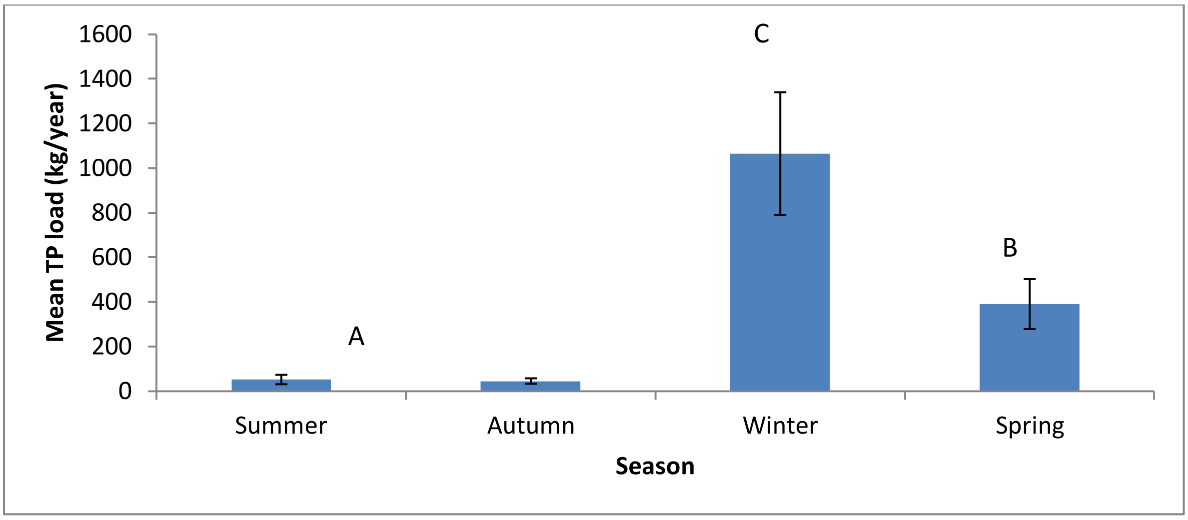

Fig. 3.28. Mean TP load (kg/year) (+/- S.E) for each season. Note: Does not include June data. Letters indicate a significant difference at α0.05.

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between the mean TP load (kg/year) and the percentage of riparian planting on each drain. There was no significant relationship found ($F= 0.2$, d.f= 9, $p= 0.7$) (Figure 3.29).

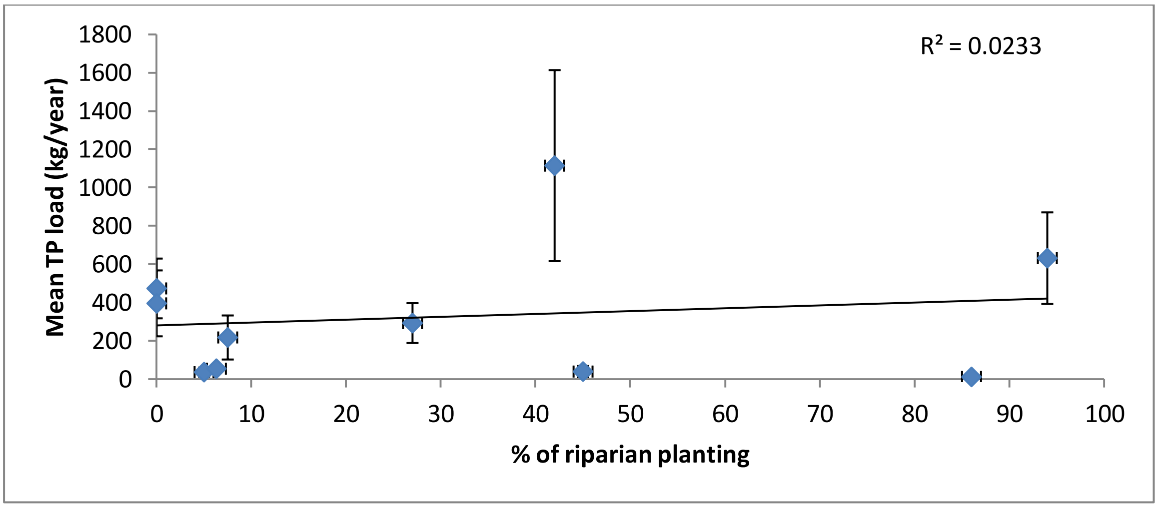

Fig. 3.29. Mean TP load (kg/year) (+/- S.E) and percentage of riparian planting on each drain. Note: excludes June data.

Flow effects

A linear model regression was conducted to test for a relationship between the average TP load (kg/day) and flow to show how flow impacts the load of P carried. There was a significant relationship found (F= 210.6, d.f= 159, p= <0.001, R^2= .7) (Figure 3.30). As flow increases so does the TP (kg/day) load.

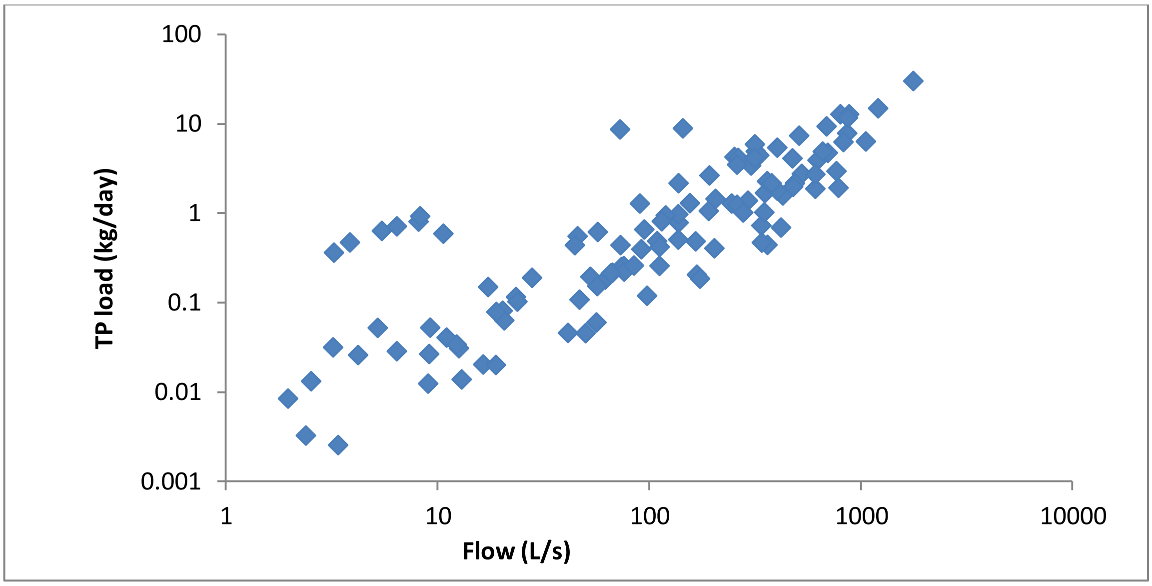

Fig. 3.30. TP load (kg/day) and flow (L/s) across all sites for entire sampling period. Note: Includes June data (graph is presented on a logarithmic scale).

3.7.2. Suspended Particulate Matter

Load values were calculated excluding data from June. The means for each site ranged between 3361 kg/year at site 7 and 194,640 kg/year at site 6 (Table 3.27). The average load of SPM per drain is 67,943 kg/year.

Table 3.27. Mean, number of samples (N), min. - max. values, standard deviation (S.D.) and standard error (S.E.) for SPM loads (kg/year) at each sampling site excluding high flow events.

Site	Mean	n	Min. - Max.	S.D.	S.E.
1 (Clarks Rd.)	626	11	0 - 4673	1497	452
2 (Embankment Rd.)	965	11	0 - 6040	2172	655
3 (Powells/Pannetts Rd.)	113554	11	0 - 363468	118659	35777
4 (Near Coes Ford)	157307	11	0 -734151	238145	71803
5 (Wolf Creek)	47107	11	0 - 280529	86707	26143
6 (Hanmer Rd.)	194640	11	0 - 716516	241582	72840
7 (Colletts Rd.)	3361	11	0 - 11284	3629	1094
8 (Drain Rd.)	78847	11	0 - 192587	83485	25172
9 (Tramway Rd.)	67841	11	0 - 283380	84960	25617
10 (Off Lower Lake Rd.)	15182	11	0 - 49531	15303	4614

Figure 3.31 shows the SPM loads (kg/month) for each drain over the year excluding data from June.

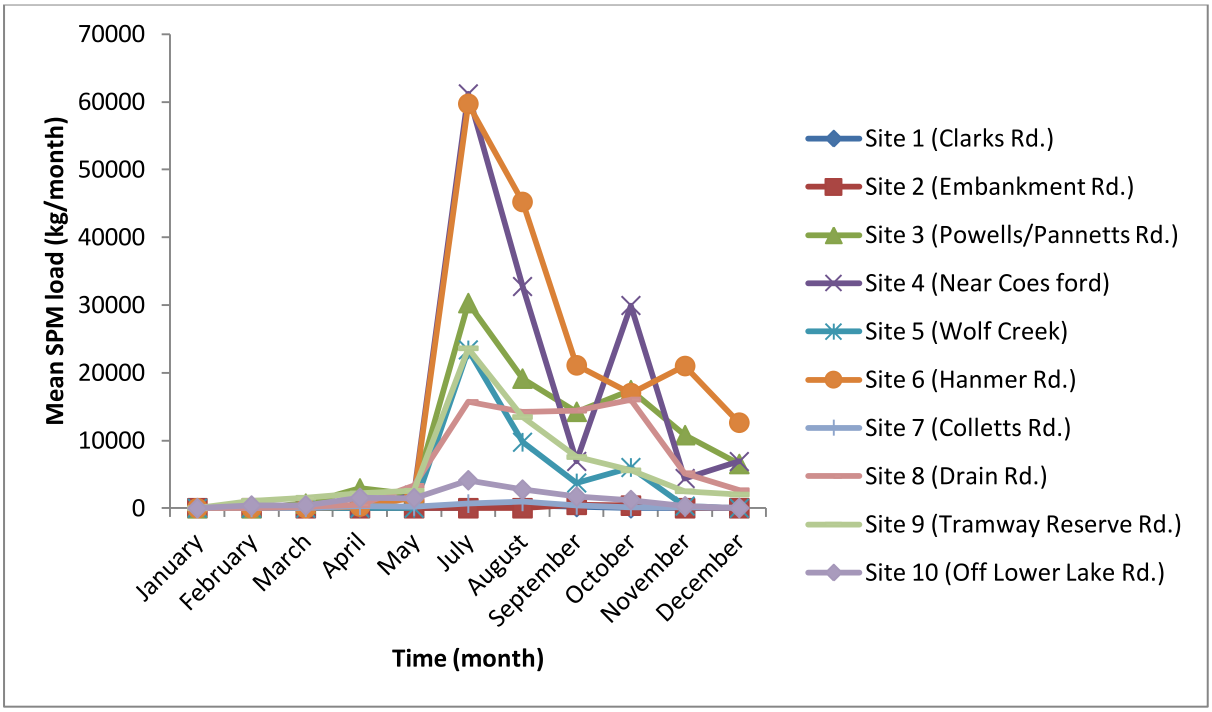

Fig. 3.31. Mean SPM loads (kg/month) for each month. Note: does not include June data.

A single factor anova was carried out on flow data to see if there were any differences in loads between drains. There was a significant difference for α0.05 ($F= 3.4$, $d.f= 109$, $p = <0.001$) (Figure 3.32). A LSD test was carried out to see which drains were different. At α0.05 sites 6 and 4 did not differ but site 6 was significantly higher than all other sites. Sites 1, 2, and 7 significantly loaded the least amount of SPM.

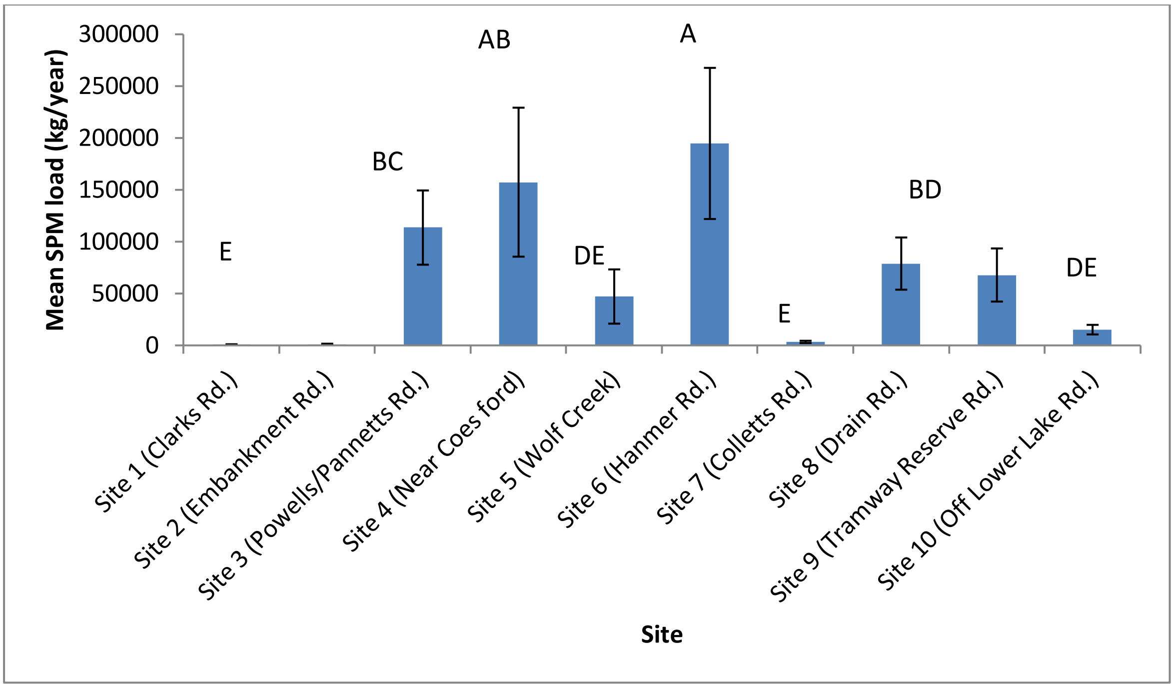

Fig. 3.32. Mean SPM loads (kg/day) (+/- S.E) for every site. Note: does not include June data. Letters indicate significant differences at α0.05.

Seasonal effects

A single factor anova was carried out on data (excluding June data to test for differences in SPM loads (kg/year) across the seasons. There was a significant difference in mean SPM loads between the seasons at α0.05 ($F=16.4$, d.f = 109, $p=<0.001$) (Figure 3.33). A least significance difference test was carried out to see which seasons significantly differed. At α0.05 significance, winter significantly loaded the most SPM, spring loaded more than summer and autumn. Summer and autumn did not differ from each other.

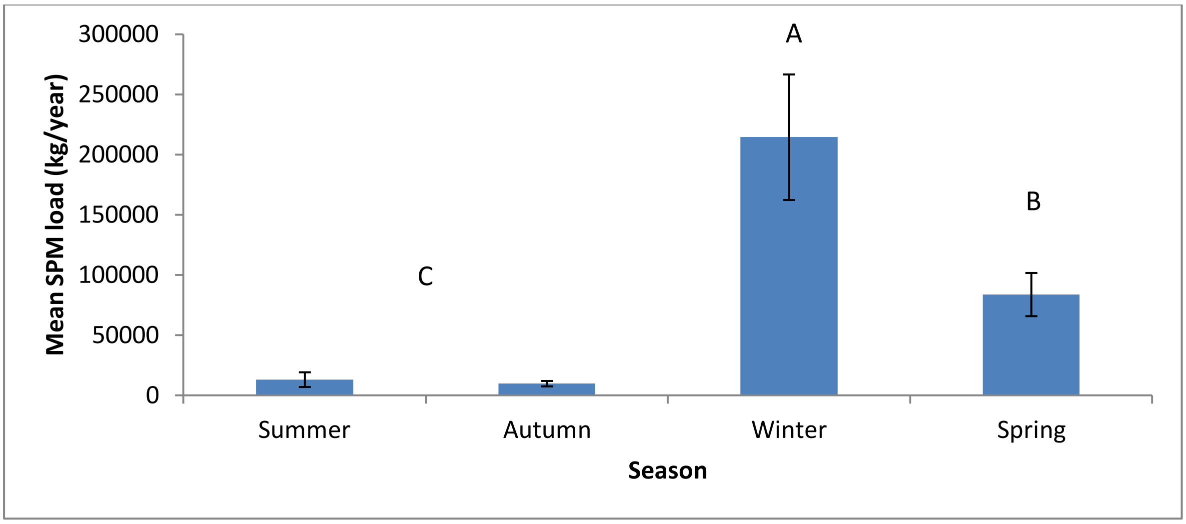

Fig. 3.33. Mean SPM load (kg/year) (+/- S.E) for each season. Note: Does not include June data. Letters indicate a significant difference at α0.05.

Riparian planting vs effects

A linear model regression was conducted to test for a relationship between the average SPM load (kg/day) and the percentage of riparian planting on each drain. There was no significant relationship found (F= 0.8, d.f= 9, p= 0.4) (Figure 3.34).

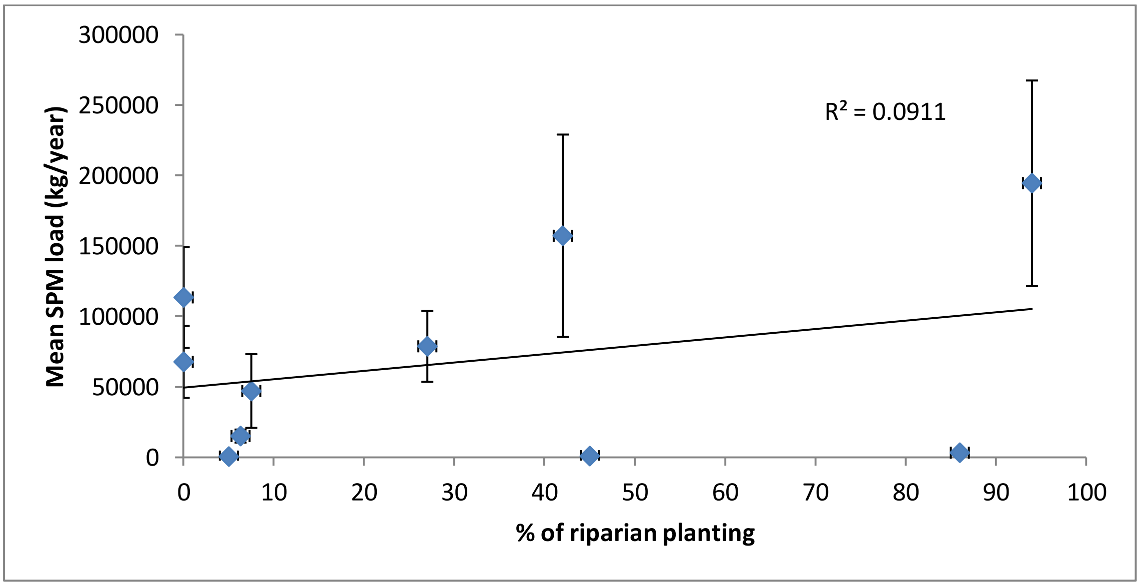

Fig. 3.34. Mean SPM load (kg/year) (+/- S.E) ad percentage of riparian planting on each drain. Note: does not include June data.

Flow effects

A linear model regression was conducted to test for a relationship between the SPM loads (kg/day) and flow to show how flow impacts the load of SPM carried. There was a significant relationship found (F= 1820.8, d.f= 154, p= <0.001, R^2 = 0.92) (Figure 3.35). As flow increases so does the SPM (kg/day)load.

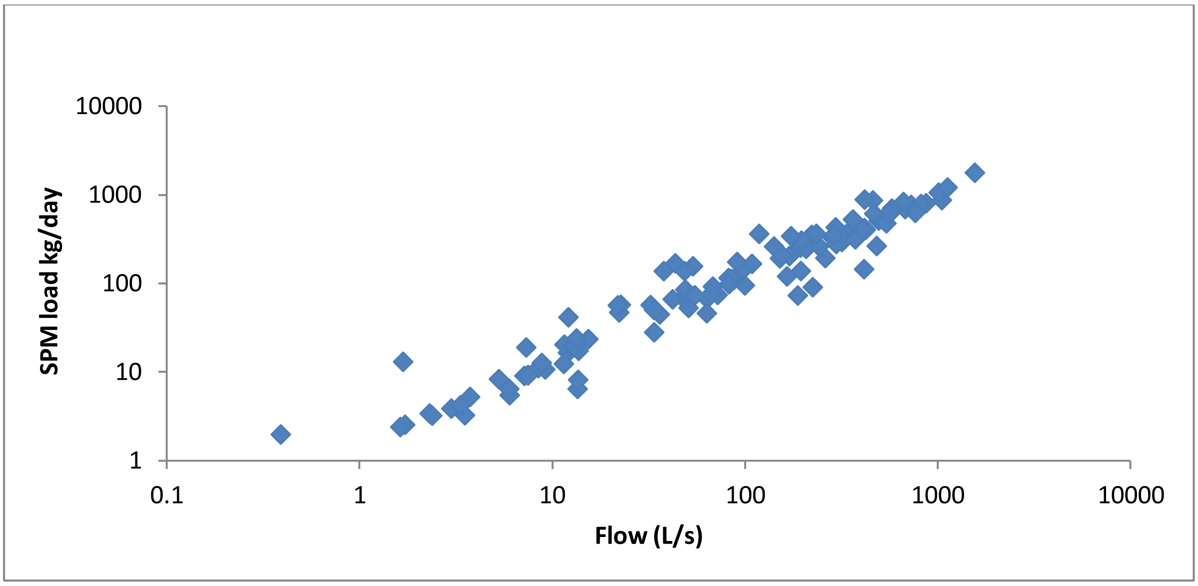

Fig. 3.35. SMP load (kg/day) and flow (L/s) over all sites and entire sampling period. Note: Graph is presented on a logarithmic scale.

Chapter 4: Discussion

The aim of this study was to determine the relationship between SPM concentrations, phosphate concentrations, ecological state and the degree of riparian restoration on drains that flowed into LE/TW, and to calculate the load of phosphorus and SPM delivered by each of the drains to LE/TW over the year, and to compare this to the loads carried by larger, natural streams and rivers.

4.1 Physical drain characteristics and riparian cover

The ten agricultural drains in this study had varying degrees of riparian drain management, with very few having purposefully planted riparian cover. This is not an unusual finding, as both within New Zealand and internationally, the intensification of farming practices usually goes hand in hand with farmland tree decline (Fisher *et al.*, 2010). Common planting that does occur on pastoral land, however, are the shelter belts which are used to protect livestock and crops from the wind. As a result, many of the drains in the LE/TW catchment, and in this study, have shelter belts planted along the drains. This has resulted in the riparian planting used in this study to only average approximately 2m wide. The widest riparian buffer width observed was approximately 4m, and that was for a purposefully planted riparian zone. None of the sites has complete shade cover, as none had closed canopies. This is not possible on many drains, due to roads running alongside of them.

Substrate types found in the drains covered 3 groups; silt/sand/mud/impervious (S/S/M/I), silt/gravel (S/G) and cobbles (C). S/S/M/I made up the substrate of 3 drains (sites 1, 2 and 5), S/G made up the substrate of 3 drains (sites 3, 4 and 7) and cobbles made up the substrate of 4 drains (sites 6, 8, 9, 10). Substrate type found at a site may be correlated to water flow, as sites 1 and 2 had low flow and site 5 did not have any flow until the extreme high flow event in June.

The physical surroundings (i.e pasture type) may also impact each of the drains differently. Dairy, sheep and deer farms all have different management strategies and different impacts on the physical aspects of the drains (i.e amount of suspended sediment and alterations to bank morphology if stock is allowed access). A study by Belsky *et al.* (1999) looked at the impacts that grazing stock can have on waterways in the Western United States. No benefits to water quality were found, instead they found decreased bank stability, increased sediment concentrations, disruption of riparian planting and an increase in peak flows due to bank compaction.

Water flow in the drains is highest in winter, second highest in spring and lowest in summer and autumn. This is attributed to higher rainfall and increased evaporation rates in summer. Flow data also shows that there is variable flow/discharge across the 10 sites. The wider and deeper drains have larger flows than the smaller, shallower drains. Site 6, for example, was approximately 2.5m wide and had a mean flow of 411 L/s, whereas site 7 was only about 1m wide and the mean flow was 9.7 L/s.

4.2 Drain water quality and ecology assessment

The Australian and New Zealand Environment and Conservation Council (ANZECC) includes ‘trigger values’ which are intended primarily to assess the risk of adverse effects to aquatic ecosystems (ANZECC, 2000). These ‘trigger values’ can be used to assess water quality. All pH means (except those of site 9 and 7) are within the minimum and maximum ANZECC guidelines (7.2 - 7.8). Site 9 is just below with a mean of 7.1 and site 7 has a mean of 6.9.

There is much debate about these guidelines being too stringent however, and a range of 6.0 - 8.5 is more suitable (James, 1999). DO means are well below the ANZECC guidelines in sites 1 and 2, with saturation levels of 98 - 105% desired for aquatic ecosystem health (around 8-9mg/L for the temperatures observed in the drains). Site 6 also had relatively low mean DO saturation (around 60%). All of the drains have higher mean TP concentrations than is stated in the ANZECC guidelines. Their guidelines state that TP should not exceed 0.03 mg/L. The lowest TP concentration mean was 0.04 mg/L at sites 7 and 10. ANZECC also state that TP levels of 0.8 - 1.2 will have negative short term effects on stream health. Site 1 always had TP concentrations >0.8mg/L and site 2 ranged from 0.71 and 1.3 mg/L.

Total phosphorus concentrations were relatively constant throughout the year. Site 1 on Clarks Road and site 2 on Embankment Road had significantly higher total phosphorus concentrations than the other drains. Observations of these two drains during the sampling period included low flow, low dissolved oxygen, high conductivity and cows in the drains, including the bodies of animals that had died, and were left in the drains to decompose (site 2). Low flows, high evaporation and direct contamination by cows may account for the higher concentrations of phosphorus.

SPM concentrations were also relatively constant throughout the year. Again, site 1 on Clarks Road, site 2 on Embankment Road, but also site 5 Wolf Creek, had significantly higher SPM concentrations than the other drains. Observations of these three drains during the sampling period include low flow and DO (especially site 1 and 2) and cows and sheep in the drains, including bodies of animals that had died, and were left in the drains to decompose (site 2 and 5). Low flows and stock having direct access to the drains may account for the higher concentrations of SPM.

Fish were never observed in drains 1, 2 or 5 but were observed in all other drains. The high levels of suspended sediment, low dissolved oxygen levels and low flow rates may account for this. A study conducted by Richardson and Jowett (2001) looking at sediment impacts on fish in New Zealand streams found that increased sediment loads do have negative impacts on fish communities (especially species that are visual predators). A study conducted in the agricultural landscape of Otago, New Zealand, found water quality declined with low flow due to less dilution of suspended sediment (Caruso, 2002). Less desirable and less diverse macroinvertebrate communities may also be a reason for the lack of fish in these drains.

Results show that the type of substrate found in a drain affected the macroinvertebrate communities. The drains with higher SPM concentrations were also the drains that had silty, sandy and muddy substrates (sites 1, 2 and 5). In these drains, the EPT% was significantly lower than in the drains that had gravel or cobble substrates because invertebrate communities were dominated by pollution tolerant taxa. If adequate riparian planting and stock exclusion was to occur along the drains, the substrate of the drain may become less silty/sandy/muddy, and better invertebrate communities may become more prevalent. Surrounding pastoral management has also been shown to impact invertebrate communities in a study conducted by Magbanua *et al.* (2010), with conventional farming practices (e.g. inputs of industrial chemicals) having more negative impacts on macroinvertebrate communities than practices that do not (e.g organic farms).

Sites 1, 2 had low DO and high TP, lack of flow and extremely high conductivity, and (with) Site 5, higher SPM concentrations. All these factors are consistent with the lack of ecology occurring in these drains, MCI and SQMCI scores are significantly lower at sites 1, 2 and 5, than in the other drains and no fish were ever observed in the water.

Site 10 had the overall best water quality out of all the drains in this study. It had the lowest (equal to site 7) TP concentrations, only slightly above ANZECC guidelines. It also had a relatively high mean DO concentration, mean pH is within the ANZECC guideline range, conductivity is low and it has the lowest mean SPM concentration out of all the sites. It has relatively (within this study) high MCI and SQMCI scores, and has fish present in the drain. It is one of the smaller drains, and it runs through a deer farm just before the sampling point. This suggests that cows and sheep may have much higher impacts on our waterways than deer do, although it has been noted that this is not an intensely studied area, despite New Zealand having a large deer industry (Klein *et al.*, 2003).

All water chemistry parameters showed significant differences between seasons except conductivity which is more influenced by catchment nature and the degree of evaporation occurring in low flow drains (e.g site 1 and 2). Mean water temperatures and pH were higher in summer and lower in winter, while mean DO levels were higher in winter (and spring) and lower in summer (and autumn). These trends likely reflected the inverse relationship between temperature and DO, and photosynthesis in summer increasing pH.

4.3 Effects of riparian cover

There was no significant relationship between total phosphorus or SPM concentrations and percentage of riparian planting, nor between percentage of riparian planting and TP or SPM loads. This indicates that, in the LE/TW catchment, the percent of a drain that has riparian planting does not impact TP or SPM concentrations in the water, or how much is transported to the lake. This finding is unexpected, but may reflect the type of riparian planting (usually “accidental”) or that dissolved reactive phosphorus made up a large percentage of total phosphorus entering the lake. Studies have shown that riparian planting may not be as efficient at removing dissolved phosphorus as removing total phosphorus (Collier *et. al*, 1995; Daniels and Gillium, 1996). Better infiltration capability of riparian buffers will improve soluble nutrient removal (Muscutt *et. al*, 1993; Collier *et. al*, 1995 , Smith, 1987).

Results also indicate that the percentage of riparian planting on a drain did not correlate with any of our macroinvertebrate diversity. A factor that must be kept in mind when looking at macroinvertebrate communities, is their ability to disperse and re-colonise. Adequate habitat is needed across both space and time to enable this dispersal. So, even if a drain has adequate riparian planting along its banks, biotic restoration may not automatically occur. This concept has been discussed by Palmer (1997) and has been named the ‘’field of dreams hypothesis’’ in which organisms will automatically colonise habitat that has been created.

The interpretation of results may also be affected by study design. The lack of riparian planting along the drains in the catchment meant that flexibility was needed when defining “riparian planting”. As explained in the methods section, shelter belts were included as riparian planting. Secondly, with no water quality data available for drains before any riparian planting occurred, there is no baseline data to compare it to. Thirdly, although comparison are being made between water quality and percentage of riparian planting between drains, each drain may not have the same amount of phosphorus leaching into the drains due to different catchment land use and nutrient management practices of the surrounding pastures. This could confound results.

Therefore, when assessing the effectiveness of riparian planting there are many variables that must be considered. Rigorous planning and enforcement of adequate riparian zones is vital to the success, of a riparian zone, which is not always the case in the LE/TW catchment. Other variables include buffer width, buffer length, adequate fencing to exclude stock and canopy cover.

4.3.1. Width of riparian plantings

The widths of riparian planting that occurred across the 10 drains ranged between 0 - 4 metres (max at site 4). There is much debate in the literature about the best width for a buffer to reduce temperature, nutrients and sedimentation, and increase invertebrate abundance and diversity. A New Zealand study by Parkyn *et. al* (2005) stated that a width of 15 -20m was needed to enable plant regeneration and to control weed growth. However, as a general rule, at least 10m is needed to be effective (Fennessy and Cronk, 1997, Stewart *et. al*, 2001; Liu *et. al*, 2008).

Multi level buffer strips are also seen as very important when seeking multiple benefits from riparian planting (Borin and Bigon, 2001; Correll, 2005). For example, the United States Forest Service has very rigorous recommendations for riparian planting including three zones to a riparian buffer. The first zone needs to be 4.5 metres wide, the second zone about 18 metres long and the third zone around 6 metres long, with each zone carring out specific functions (Correll, 2005). Most of the drains in this study did not have this tiered system in their riparian planting.

Therefore, inadequate buffer widths along the drains are a potential factor contributing to the lack of correlation between percentage of riparian planting and water quality results in this study.

4.3.2. Length of riparian planting

In this study, the percentage of riparian planting along 1000m upstream of each sampling point was used. For a majority of the drains there was not continuous planting along the banks. This could contribute to the results of this study. In an ideal world, riparian buffers would begin at the headwaters and advance downstream (Parkyn *et al*., 2005) but this is not a realistic goal due to private land ownership.

Buffer length plays a huge role in the ability of nutrient uptake to occur. It has been shown that at least 300 metres is needed for this to occur (Parkyn et. al, 2003). If there are gaps present along the riparian corridor, this enables sediment and nutrients to enter the waterways. Also, the deeper and wider the stream, the longer the buffer zone needs to be. Parkyn *et. al* (2003) said that 1-5 km for headwaters compared to 10-20km for fifth order streams with 75 % shade was needed to achieve a 5 °C water temperature reduction. It has also been known that DO levels are heavily impacted by temperature. Our results show higher DO levels in winter and spring, most likely impacted by cooler water temperatures and increased water flow. Increased DO levels would be beneficial for stream biota.

Differences in seasonal temperature across the drains indicate that there may not be adequate shading occurring from riparian planting to facilitate cooler, constant water temperatures, although temperature did not exceed safe ecological levels. Full canopy cover was not achieved at any of the sites in this study. Until more shading is present on the drain channels, we should not expect to see reductions, or stability, of water temperatures in the drains.

4.3.3. Fencing to exclude stock

As mentioned previously, 3 sites in this study regularly had animals (cows and sheep) present in the drains. Results showed that these 3 sites (sites 1, 2 and 5) also had significantly higher SPM concentrations than the other drains. Fencing off waterways from stock access has positive outcomes for water quality, as stock cannot harm habitat, defecate in the water or trample the banks which leads to less erosion (Muscutt *et. al*, 1993; Collier *et. al*, 1995, Smith, 1987). Eliminating stock access could also reduce dissolved nutrients from entering waterways, as less compacted ground from trampling would allow for better nutrient filtration into the soil.

However, out of the 3 sites that had animals in the drains, two (site 1 and site 2) did actually have fencing. The issue was that the land owners did not ensure that the animals stayed inside the fenced properties. This highlights that some farmers may not realise the importance of fencing out stock. A study conducted by Bewsell *et. al* (2007) asking New Zealand dairy farmers why they fenced out stock from their waterways, highlighted that fencing was usually undertaken to manage stock. Farmers generally didn’t mention improvements in water quality as a reason to fence.

4.4. Drain contribution to Lake Ellesmere/Te Waihora

One of the aims of this study was to calculate the loads of phosphorus and SPM entering Te Waihora via the drains. It is important to distinguish the amount that is actually discharging into the lake, rather than just consider the concentrations that are occurring in the surrounding tributaries. For instance, the results of this study showed that the drains which have relatively low concentrated levels of phosphorus and SPM, are contributing more phosphorus and SPM to the lake due to their higher flow regimes. Site 6, for example has statistically lower total phosphorus and SPM concentrations than 3 other drains, but it is one of the drains that loads the most TP and SPM into the lake. This is because site 6 drain has one of the highest mean flow, and is also one of the largest drains.

Seasonal differences in total phosphorus loads indicate that the loads are highest in winter, when flows are also highest. As flow increases, so does the TP loads. The average load (per drain) of total phosphorus was calculated to be 326 kg/year. This number multiplied by 80 drains (the approximate number of drains loading into the lake as observed on Google Earth) gives an approximate total phosphorus load entering the lake via the drains at 26, 080 kg/year.

When TP concentrations are relatively high (>0.1 mg/L), dissolved reactive phosphorus contributed up to 100% of the load with a mean of 69% of the TP. This is somewhat higher than has previously been assumed, and indicates a relatively important role for dissolved P transport contributing to the lakes phosphorus load. Larned and Schallenberg (2006) estimated the ungauged tributaries of LE/TW had mean TP concentrations of 0.06 mg/L, 50% of which would be made up of DRP (0.03 mg/L). The mean for TP concentrations across all of the drains in this study was 0.28 mg/L but that is affected by the extremely high concentrations at sites 1 and 2. If sites 1 and 2 TP concentrations are excluded, the mean is 0.07 mg/L which is only slightly above Larned and Schallenbergs estimate. The DRP% from this study however is still greater than the estimated percentage.

SPM loads carried by the drains were significantly higher in winter, compared to the other seasons. Once again, this is due to higher flow occurring in winter as SPM concentrations were relatively constant over the year. The mean load (per drain) of SPM was calculated to be 67,943 kg/year. This value multiplied by 80 drains (the approximate number of drains into the lake, as observed on Google Earth) gives a total approximate load of SPM entering the lake of 5435 tonnes/year.

4.5. Drain contributions compared to larger tributaries

In the following discussion all river data referred to is from an ECan monitoring data set provided courtesy of Environment Canterbury. Using monthly monitoring data from ECan, comparisons were made between TP and SPM loadings from agricultural drains and from the larger, more natural rivers that flow into Te Waihora. This was to determine how much and where the key TP and SPM loads to the lake were sourced from. River data was compiled from the Selwyn, Halswell and L2 Rivers. Figure 4.1 shows the mean TP load (kg/year) per river and per drain.

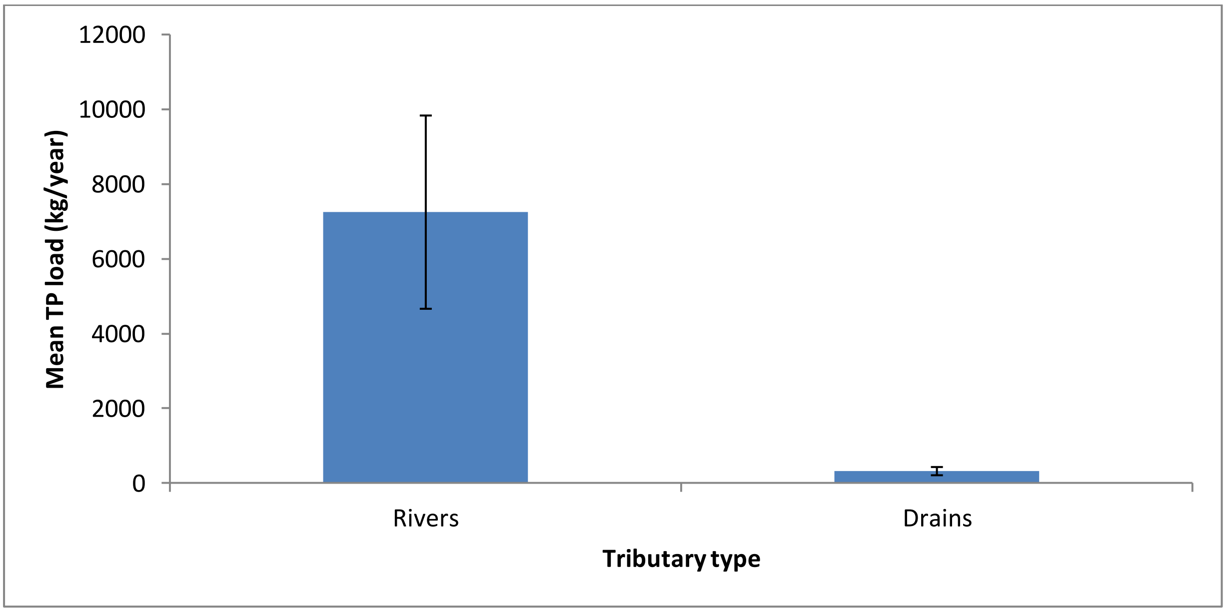

Fig. 4.1. The mean TP load (+/- S.E.) per tributary type (i.e per river and per drain).

Figure 4.2 shows the total load of TP from each tributary type, calculated as the mean annual TP load for each river, multiplied by 5 (number of major rivers that flow into Te Waihora: Selwyn, L2, Halswell, Kaituna Rivers and Harts Creek) and the mean TP load for each drain was again multiplied by 80 to determine total drain input.

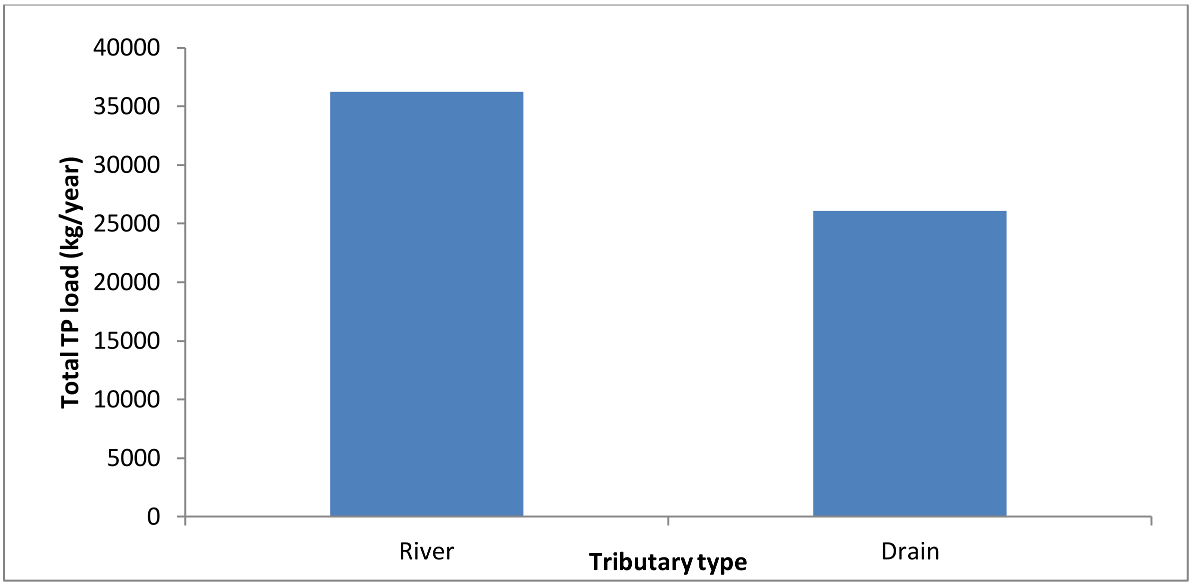

Fig. 4.2. Total TP loads (kg/year) for rivers and drains.

The 5 major rivers contribute an estimated 36, 260 kg/yr of TP to LE/TW. The smaller tributaries or drains contribute approximately 26, 080 kg of TP/year to Lake Ellesmere.

Both the drains and the rivers had higher than previously assumed loads of dissolved reactive phosphorus being loaded into the lake, drains had on average have 69% of TP present as DRP, whereas the rivers had 60% of TP present as DRP. This highlights the need for restoration work to focus on removing dissolved P from the drains as well as sediment-bound phosphorus.

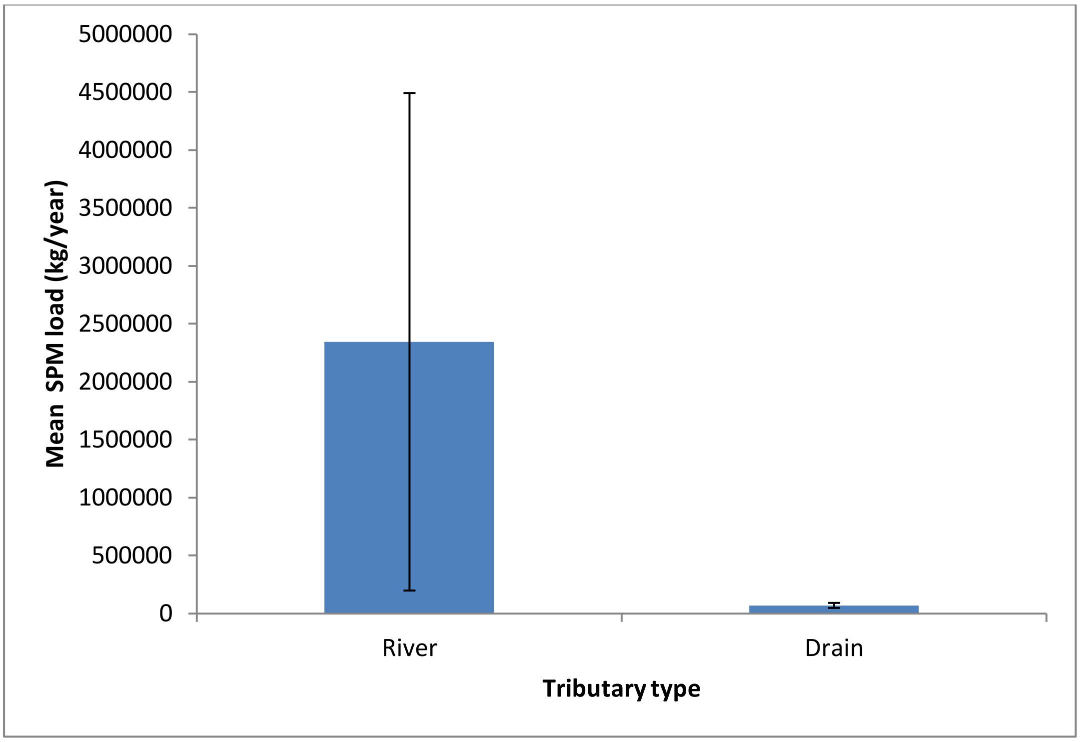

Fig. 4.3. Mean SPM loads (+/- S.E.) per tributary type (i.e per river and per drain).

Figure 4.3 shows the mean SPM load (kg/year) per river and per drain and Figure 4.4 shows the total load of SPM (tonnes/year) from each tributary type. To derive the latter, the mean SPM load for each river was multiplied by 5 (number of major rivers that flow into LE/TW) and the mean SPM load for each drain was multiplied by 80.

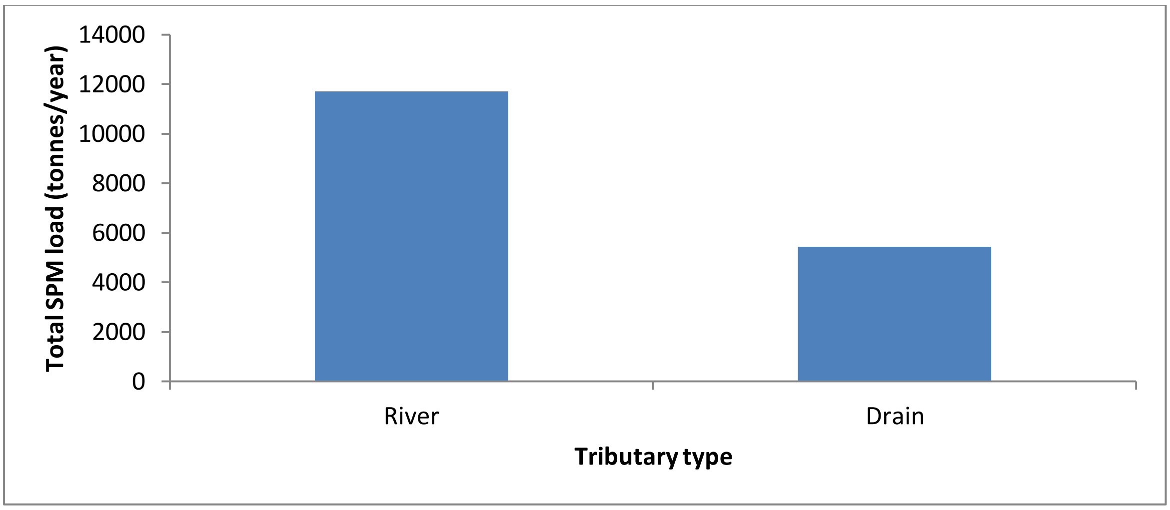

Fig. 4.4. Total SPM load (tonnes/year) for rivers and drains.

In 2013, the 5 major rivers contributed an estimated 11,719 tonnes of SPM to LE/TW annually. Drains contributed approximately half as much at 5,435 tonnes of SPM annually to LE/TW.

Therefore, despite the fact that the TP and SPM concentrations found in the drains were higher than those of the larger rivers, due to their larger catchments and flows, the rivers carried larger loads into the lake. None-the-less the contribution from the drains was still very significant.

4.6. Reducing drain phosphorus loads

If reduced phosphorus and SPM loads are a desired outcome for the LE/TW catchment, then more extensive and effective riparian planting is needed. Implementation can be complicated however. As Parkyn (2003) stated, riparian planting should start in the headwaters and continuously progress down to the lower water bodies, but this is unrealistic. One possible solution is to consider riparian planting that can double as shelter belts. Farmers understand the need for shelter belts on their farms as a way to protect stock and their land. If these shelter belts could be more modified to double as protection for the drains that run along their pastures, it could be a win-win situation. It highlights the issue that farmers need to understand their impact that their practices are having on the environment. The study by Beswell *et. al* (2007) shows that farmers tend to worry more about their animals rather than worrying about the impacts that they are having on the surrounding water bodies. However, planting can be very expensive so expecting the farmings to deal with these issues alone may not be the answer.

The higher than expected dissolved P concentrations in both the major rivers and the drains, shows the need for restoration efforts targeting reduced dissolved P in the waterways. This can be achieved through better riparian management (more water and nutrient infiltration into the soil) but could also be achieved through the introduction of in-stream mechanisms. It has been shown that having macrophytes and periphyton in streams can reduce P loads (Pelton *et. al*, 1998; Dodds, 2003) by encouraging P uptake and deposition. It needs to be noted however that usually macrophytes need to be removed after a few years as they can become saturated in P which can lead to higher fluxes being released during die off (Richardson, 1985).

Restoration attempts could also focus on high flows in winter and spring which results in greater TP and SPM loads. Phosphorus uptake has been shown to negatively correlate with stream velocity (Reddy *et. al*, 1999); the higher the stream velocity, the less P retention occurs. One way that water flows might be reduced is by introducing debris dams into the drains. This would alter the morphology of the drains, possibly slowing down the flow and allowing more time for P uptake to occur (Reddy *et al*., 1999). The dam could also act as a sediment buffer and encourage more course particulate organic matter (CPOM) in the streams. Aldridge *et.al* (2009) looked at the introduction of CPOM in streams. It increased microbial activity and phosphorus retention due to higher surface area availability for attachment. CPOM may also provide energy inputs into food webs, thus increasing biological diversity and stability, in addition to retaining more P to fuel further growth. Once again, these debris dams would also need on-going management as the sediment would need to be removed.

4.7 Limitations of this study

One limitation of this study may have been the inclusion of shelter belts as “riparian planting”. In future research, a better assessment of the quality of riparian planting should be considered. Although, this may be difficult, due to the lack of riparian planting on the drains around LE/TW, especially along the waterways that have not already been extensively studied (e.g. Collins *et al*., 2012).

More in-depth research could also be undertaken on the relationships between water quality parameters and macroinvertebrate community structure. Only the percentage of riparian planting and substrate type was investigated in detail due to the scope of the study, but other parameters such as dissolved oxygen, contaminants, temperature and flow may impact macroinvertebrate communities.

The methodology used, in this study, for analysing dissolved reactive phosphorus (DRP) was also a limitation as it was not sufficiently reliable at low TP concentrations.

Chapter 5: Conclusions

The aim of this study was to determine the relationship between SPM concentrations, phosphate concentrations, ecological state and the degree of riparian restoration on drains that flowed into Lake Ellesmere/Te Waihora, and to calculate the load of phosphorus and SPM delivered by each of the drains to LE/TW over the year, comparing this to the loads carried by larger, natural streams and rivers. The conclusions of this study are as follows;

- Sites 1, 2 had low DO and high TP, lack of flow and extremely high conductivity, and (with) Site 5, higher SPM concentrations. All these factors are consistent with the lack of ecology occurring in these drains. All drains failed to meet ANZECC guidelines for TP concentrations. All water chemistry parameters showed significant differences between seasons except conductivity. Mean water temperatures and pH were higher in summer and lower in winter, while mean DO levels were higher in winter (and spring) and lower in summer (and autumn).
- The degree of riparian planting present on a drain did not appear to affect total phosphorus, suspended particulate matter, macroinvertebrates or general water chemistry in the drains. However, limited purpose- planted riparian planting on the drains in the catchment, and the decision to include shelter belts as riparian planting, may have influenced these results. Not all riparian planting is of equal effectiveness, and these results may reflect that.

- Macroinvertebrate analyses indicate moderate to severe pollution in all the drains despite the amount of riparian planting present. Macroinvertebrate community diversity appeared to be related principally to substrate type, with silty, sandy, muddy and impervious substrates resulting in pollution tolerant taxa dominating the community.

- Despite the high concentrations of phosphorous and SPM found in low flow drains (e.g., sites 1 & 2), most of the phosphorus and suspended particulate matter delivered to the lake is from the larger (wider and deeper) higher flow drains, and occurs in winter and spring. The farm drain tributaries of LE/TW are estimated to deliver 26, 080 kg/yr TP and 5,435 tonnes/yr SPM to the lake.
- When compared to the contributions of the larger, natural rivers over the same period (from ECan monitoring data), the drains contribute approximately 50 to 70% as much as the larger waterways.
- Dissolved reactive phosphorus (DRP) made up 69% on average, of the total phosphorus in the drains; a value that was higher than had previously assumed in both the drains and the larger, natural rivers and streams.
- To be most effective, restoration work should therefore be aimed at preventing sediment and phosphorus entering the larger drains in winter and spring. More adequate riparian planting needs to occur on these drains, and it should be managed in a way that a reduction in dissolved phosphorus, as well as in sediment and sediment-bound phosphorus, can be achieved.

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Appendix

NA= Data not available
NW= No water.
January data is not presented as no samples were collected.

1. Water Chemistry raw data

1.1. Dissolved Oxygen (mg/L)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	0.47	NW	3.96	NW	NW	2.95	3.33	4.61	7.28	5.9
20.03.2013	1.2	1.45	2.73	NW	NW	8.93	6.44	7.12	7.98	8.04
22.04.2013	NW	0.23	3.45	4.43	NW	8.6	5.02	7.02	7.02	9.09
27.05.2013	0.28	0.48	6.41	8.83	4.11	10.38	4.97	10.47	9.3	9.67
18.06.2013	4.85	4.47	6.8	NA	8.8	11.08	5.98	10.25	8.59	8.63
9.07.2013	6.16	3.02	10.23	10.68	9.19	11.72	5.7	11.27	NA	NA
22.07.2013	5.34	0.27	9.31	9.72	8.9	10.59	7.99	10.54	9.63	9.27
12.08.2013	9.26	4.4	11.38	11.8	9.9	13.19	9.82	15.14	12.03	11.78
27.08.2013	8.87	3.8	10.33	NA	9.6	11.55	8.99	12.88	11.81	10.63
12.09.2013	10.99	8.57	12.03	12.09	14.02	12.32	8.48	14.24	11.99	10.34
27.09.2012	4.1	4.4	11.66	11.62	10.44	11.89	8.55	13.98	11.92	11.01
14.10.2013	1.6	1.8	12.04	11.33	12.1	11.88	8.56	12.71	11.45	11.04
28.10.2013	0.52	0.68	13.37	10.52	13.55	11.94	8.4	11.2	11.01	11.12
23.11.2013	0.35	0.26	12.54	10.79	9.54	11.7	4.01	10.93	9.35	10.54
3.12.2013	0.32	0.31	11.04	9.96	9.43	10.25	3.95	9.98	8.64	9.73

1.2. Water temperature (°C)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	16.5	NW	17.01	NW	NW	17.3	17.9	18.3	17.1	15.5
20.03.2013	10.8	12.2	13.1	NW	NW	12.2	13.6	14.3	17.3	12.8
22.04.2013	NW	12.1	13.4	12.4	NW	11.7	13.3	13.1	14.4	14.2
27.05.2013	8.8	8.4	11.7	8.9	8.5	9	10.6	9.5	11.5	11.8
18.06.2013	10.3	8.7	10.5	NA	5.5	7.5	8.3	7.5	8.9	11.4
09.07.2013	8.7	7.3	8.8	7.5	6.5	7.3	8.3	7.4	NA	NA
22.07.2013	8.7	10.6	10.3	10	9.8	10.1	10.5	11	11.3	13.1
12.08.2013	11.2	10.4	11.2	10.9	10.6	10.8	11	11.1	11.2	12.8
27.08.2013	10.4	9.9	10.9	NA	10.7	10.6	10.6	11.2	11.1	12.4
12.09.2013	11	10.5	10.7	11.7	11	10	10.2	11.9	12.8	12.8
27.09.2013	10.8	9.9	11	11.3	11.2	10.1	10.5	11.8	12.6	12.4
14.10.2013	14.5	12.6	11.8	11.3	11.8	10.4	11.3	12.1	12.4	12.9
28.10.2013	16.9	15.1	13.5	14.1	13.2	13.6	13.6	14.6	14.7	13.9
23.11.2013	19.1	18.6	15.8	15.6	17.9	16.7	16.3	17.6	16.1	15.2
3.12.2013	19.3	18.7	18.9	16.8	17.9	18.3	17.2	20.3	19.6	17.1

1.3. pH

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	7.2	NW	6.85	NW	NW	6.92	6.54	6.66	6.82	6.61
20.03.2013	7.1	7.18	7.08	NW	NW	7.82	7.28	7.41	7.98	7.53
22.04.2013	NW	6.9	7.44	6.94	NW	7.21	6.55	6.47	6.55	6.99
27.05.2013	7.3	7.2	6.95	6.86	6.77	7.82	6.64	6.93	6.81	7.16
18.06.2013	6.85	6.98	6.62	NA	6.56	6.72	6.67	6.8	6.88	7.04
9.07.2013	6.97	6.98	6.38	7	7.21	7.54	7.33	7.74	NA	NA
22.07.2013	6.85	6.98	6.78	6.93	6.86	7.25	6.92	7.4	7.05	7.15
12.08.2013	7.24	6.92	6.7	6.93	7.06	8.16	7.2	8.54	7.2	7.26
27.08.2013	7.11	6.95	6.74	NA	7.01	7.64	7.18	7.85	7.03	7.02
12.09.2013	7.37	7.5	6.86	7.5	7.57	7.95	6.79	8.4	7.24	7.31
27.09.2013	7.21	6.81	6.9	7.45	7.29	7.81	6.91	7.8	7.04	7.03
14.10.2103	7.25	6.9	6.87	7.38	7.33	7.54	6.99	7.11	6.92	7.13
28.10.2013	7.85	8.24	7.96	7.58	7.8	8.2	6.99	7.92	7.13	7.35
23.11.2013	7.73	8.41	8.12	7.72	7.75	9.12	7.12	8.33	7.43	7.86
3.12.2013	7.8	8.44	8.63	7.85	7.77	8.77	7.15	8.44	7.5	7.83

1.4. Conductivity (µS/cm)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	1882	NW	293	NW	NW	451	267	274	273	192.7
20.03.2013	2510	1109	357	NW	NW	510	298	354	300	218.6
22.04.2013	NW	2250	376	525	NW	507	292	325	300	241
27.05.2013	3500	2400	348	411	501	510	301	373	318	234
18.06.2013	659	330	269	NA	319	361	411	400	375	299
09.07.2013	2770	2490	315	338	338	306	302	360	NA	NA
22.07.2013	3109	2501	359	341	365	327	328	371	356	237
12.08.2013	2650	2360	285	298	347	288	287	318	321	201.2
27.08.2013	1898	2266	310	NA	351	297	298	299	301	198.6
12.09.2013	2520	2270	276	286	414	282	269	314	312	185.2
27.09.2013	2600	2210	281	301	388	299	286	344	301	193
14.10.2013	2920	2255	291	311	386	297	281	332	304	195
28.10.2013	1875	2470	263	274	391	267	255	309	312	193
23.11.2013	1893	1989	277	324	402	287	289	307	287	194
3.12.2013	1946	2200	291	380	411	345	291	301	295	199

2. Total phosphorus (mg/L) raw data

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	NA	NW	NA	NW	NW	NA	0.06	0.11	0.11	0.05
20.03.2013	NA	NA	0.03	NW	NW	NA	0.01	0.03	0.04	0.05
22.04.2013	NW	NA	0.07	0.07	NA	0.04	0.01	0.06	0.04	0.03
27.05.2013	NA	NA	0.05	0.04	0.09	0.03	0.03	0.03	0.03	0.04
18.06.2013	1.37	0.71	0.19	NA	0.22	0.20	0.05	0.07	0.14	0.08
9.07.2013	1.29	0.99	0.06	0.10	0.18	0.03	0.04	0.04	0.05	0.04
22.07.2013	1.31	1.34	0.04	0.16	0.16	0.15	0.08	0.09	0.07	0.07
12.08.2013	1.41	1.31	0.17	0.18	0.18	0.08	0.10	0.05	0.13	0.04
27.08.2013	1.28	1.16	0.18	NA	0.16	0.10	0.05	0.05	0.15	0.04
12.09.2013	1.34	1.14	0.05	0.07	0.09	0.07	0.03	0.04	0.15	0.03
27.09.2013	1.29	1.29	0.02	0.07	0.08	0.04	0.02	0.03	0.15	0.01
14.10.2013	0.64	0.64	0.06	0.17	0.08	0.09	0.01	0.05	0.02	0.01
28.10.2013	1.23	0.80	0.06	0.15	0.14	0.02	0.02	0.01	0.01	0.01
23.11.2013	1.05	1.30	0.01	0.08	0.12	0.06	0.07	0.02	0.01	0.01
3.12.2013	NA	0.89	0.10	0.12	0.11	0.06	0.05	0.05	0.04	0.05

3. DRP raw data (mg/L)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	NA	NW	NA	NW	NW	NA	0.13	0.11	NA	0.09
20.03.2013	NA	NA	0.03	NW	NW	NA	0.11	0.12	0.13	0.10
22.04.2013	NW	NA	0.01	0.05	NW	0.08	0.03	0.06	0.03	0.02
27.05.2013	NA	NA	0.02	0.04	NA	0.04	0.01	0.03	0.08	0.01
18.06.2013	0.71	0.70	0.13	NA	0.18	0.15	0.08	0.05	0.13	0.07
09.07.2013	0.65	0.58	0.07	0.15	0.13	0.06	0.08	0.05	0.08	0.03
22.07.2013	0.65	0.56	0.07	0.11	0.07	0.05	0.04	0.08	0.05	0.04
12.08.2013	0.62	0.58	0.18	0.16	0.09	0.17	0.06	0.03	0.14	0.02
27.08.2013	0.70	0.44	0.07	NA	0.26	0.05	0.04	0.06	0.08	0.03
12.09.2013	0.64	0.62	0.05	0.11	0.04	0.03	0.02	0.06	0.01	0.02
27.09.2013	0.78	0.76	0.05	0.13	0.04	0.04	0.05	0.03	0.01	0.01
14.10.2013	0.61	0.45	0.06	0.08	0.07	0.06	0.02	0.05	0.03	0.02
28.10.2013	0.90	0.90	0.06	0.11	0.11	0.07	0.02	0.03	0.03	0.01
23.11.2013	0.90	0.90	0.03	0.13	0.11	0.06	0.04	0.04	0.02	0.02
3.12.2013	NA	0.88	0.07	0.10	0.08	0.06	0.02	0.06	0.01	0.02

4. SPM raw data (g/L)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	NA	NW	NA	NW	NW	NA	0.0079	0.0086	0.0094	0.0079
20.03.2013	NA	NA	0.0055	NW	NW	NA	0.0079	0.0095	0.0112	0.0065
22.04.2013	NW	NA	0.0081	0.0094	NW	0.0089	0.0085	0.0076	0.0113	0.0076
27.05.2013	NA	NA	0.0086	0.011	0.0299	0.0093	0.0081	0.0076	0.0088	0.0067
18.06.2013	0.0499	0.0533	0.0089	NA	0.0389	0.0102	0.0066	0.0181	0.0108	0.0188
09.07.2013	0.0488	0.0544	0.0073	0.0062	0.0512	0.0211	0.0073	0.008	NA	NA
22.07.2013	0.0311	0.0432	0.034	0.054	0.027	0.034	0.014	0.014	0.021	0.017
12.08.2013	0.0134	0.0522	0.0132	0.0156	0.0364	0.0396	0.0316	0.0122	0.0075	0.0074
27.08.2013	0.0444	0.062	0.022	NA	0.0288	0.0192	0.0288	NA	0.0216	0.0132
12.09.2013	0.01976	0.0394	0.0146	0.0108	0.016	0.0172	0.0208	0.0224	0.0138	0.0166
27.09.2013	0.0126	0.0074	0.0198	0.0038	0.0083	0.0089	0.0092	0.0126	0.0063	0.0078
14.10.2013	0.0118	0.01	0.015	0.0055	0.0192	0.0093	0.0045	0.0107	0.0059	0.0095
28.10.2013	0.0312	0.0446	0.0195	0.0322	0.046	0.0115	0.0079	0.0145	0.0107	0.0094
23.11.2013	0.042	0.0526	0.0245	0.0122	0.0232	0.0308	0.0102	0.0097	0.0098	0.0085
3.12.2013	NA	0.0461	0.016	0.046	0.0244	0.0252	0.0112	0.0093	0.0102	0.0063

5. Raw flow data (L/s)

Date of sample	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
28.02.2013	0	0	0	0	0	0	2.52	3.2	44.4	18.9
20.03.2013	0	0	46.72	0	0	0	3.38	9.1	52.55	23.76
22.04.2013	0	0	137.28	9.2	0	11	16.38	23.4	73.7	75.9
27.05.2013	0	0	91.52	66.825	0	61.765	12.58	165.24	111.53	84.48
18.06.2013	72.6	143.83	252	NA	314.7	1764.34	20.25	1052.4	1203	205.06
09.07.2013	0	0	351	859.08	262.72	781.7	20.6	426.5	477.4	111.7
22.07.2013	0	0	764.55	686.75	192	866.64	27.88	658.5	377.85	72.96
12.08.2013	3.843	0	509.35	797.98	137.2	696	17.28	400.78	302.13	65.76
27.08.2013	6.405	0	317.64	NA	90.1	473.8	6.4	486.64	401.59	136.8
12.09.2013	5.44	8.11	291.94	360	119.28	622.08	12.28	277.2	312.7	56.64
27.09.2013	3.23	8.25	337.76	360	114.31	607.5	9	348.8	259.31	49.92
14.10.2013	0	10.62	524.25	877.6	94.5	825	18.81	607.76	339.44	56.17
28.10.2013	0	0	244.4	330	45.75	418.32	2.38	360.96	173.28	41.22
23.11.2013	0	0	167.48	136.8	5.2	259.2	4.2	202.58	97.4	12.96
3.12.2013	0	0	155.2	56.98	0	190.56	0	108.68	75.64	1.96

6. Macroinvertebrate raw data

Blank columns mean there was no water. Only taxa that were present for each months sample is in the table. Collection in April was not possible due to equipment malfunction.

6.1. 28.02.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Deleatidium	8								2		
Aoteapsyche	4										8
Hudsonema	6						30		20	50	
Hydrobiosis	5									1	3
Oxyethira	2			10				1			
Pycnocentria	7						30	5	3	1	18
Potamopyrgus	4	15		15			10	20	150	11	16
Gyraulus	3			30					8		
Physa	3			2					3		2
Antiporus	5										1
Chironomidae	1	60									
Tanypodinae	5						2				
Sigara	5						1				
Ostracoda	3						200	20	50		
Amphipoda	5			15						15	10
Dolomedes	5									1	
Oligochaeta	1						2	30	120		
Nematoda	3								15	15	

6.2. 20.03.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Hudsonema	6								8	1	5
Oxyethira	2			12							10
Psilochorema	8										1
Pycnocentria	7									15	
Physa	3	2	6	15			8		3		
Tanypodinae	5	3					15				10
Chironomidae	1	10	15	15						30	25
Ostracoda	3			25			10		25	10	
Amphipoda	5			10				15			
Sphaerium	5			5							
Dolomedes (Spider)	5										1
Platyhelminthes	3							3			
Oligochaeta	1		3				4		15	25	
Nematoda (horsehair)	3							3	8	20	

6.3. 22.05.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Hudsonema	6				2		15	3			
Oxyethira	2			10				8			
Psilochorema	8										2
Pycnocentrodes	5								30	40	
Triplectides	5						10		30	20	10
Potamopyrgus	4	25	20		30		25	20	120	10	50
Physa	3					25	10	5	3		
Sphaeriidae	3								3		
Chironomidae	1	8	10	100	20	20			30		
Austrosimulium	3					10					
Tipulidae	5			50							
Ostracoda	3		25	20			80	20		20	10
Amphipoda	5						20			80	300
Oligochaeta	1	5		3	1	3	5		30	25	
Nematoda (horsehair)	3							10	15	30	

6.4. 18.06.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Zelandobius	5				1				3	5	
Deleatidium	8						5	1	3		
Hudsonema	6			15							
Hydrobiosis	5						15				
Oxyethira	2		10	8	3	20	10	8	15	6	5
Psilochorema	8										2
Triplectides	5						10			3	
Potamopyrgus	4	20	50	80	30	25	25	50	30	20	10
Physa	3	8	2		5	25		8	5	4	
Sphaeriidae	3						15		10		
Elmidae	6						10			2	
Chironomidae	1	30	150	120	30	100	110	45	30	50	20
Tanypodinae	5	8	25	12							
Ostracoda	3			25	25		30			40	
Amphipoda	5							12	25		25
Oligochaeta	1	4	5	10	8			3	3		

6.5. 28.07.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Deleatidium	8			1	6		3	3	2		
Hudsonema	6								20		
Oxyethira	2							5			
Psilochorema	8				1						
Pycnocentrodes	5			3						20	
Triplectides	5						28		30	80	25
Potamopyrgus	4	50	30	15	5			10	20	15	30
Lymnae	3	30		10					30	10	
Physa	3	3	25			12	15	6	5	30	
Sphaeriidae	3						2			3	
Chironomidae	1		30	10		15	25				15
Austrosimulium	3				3						
Tanypodinae	5		10	15					30		
Ostracoda	3	5		15	3	15	10		10	20	
Amphipoda	5			25						50	20
Oligochaeta	1	25	30	10		3	23	5	5		
Nematoda (horsehair)	3		5				12	10			

6.6. 27.08.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Deleatidium	8			25		1		1			
Oxyethira	*2*		3					5			3
Psilochorema	*8*									2	2
Pycnocentrodes	*5*						3			40	
Triplectides	*5*			20			30			10	
Potamopyrgus	4	10	150	200			15	100			5
Lymnae	3									10	
Physa	3			5		8	80			5	
Sphaeriidae	3									3	
Chironomidae	1	80	100	150		10	5				
Tanypodinae	5										120
Ostracoda	3		8			120	20				
Amphipoda	5			50		80		10		50	100
Oligochaeta	1	4					10			10	
Platyhelminthes	3							1			

6.7. 27.09.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Zelandobius	5							1			
Deleatidium	8			10	3			2	1	1	
Hudsonema	6				10					30	3
Oxyethira	2					20			3	15	10
Polyplectropus	8				1		1			1	
Psilochorema	8				1					1	1
Triplectides	5			30			12				
Potamopyrgus	4	45	30	30	30	10	20	30	20	20	15
Physa	3							3	15		3
Sphaeriidae	3					30	30				
Elmidae	6						10	4	5		
Chironomidae	1	80	100	100	60	80	100	50	150	50	30
Austrosimulium	3	25									
Tanypodinae	5		70		40						
Ostracoda	3			30	25				100	50	
Amphipoda	5			20						100	15
Copepoda	5							50			
Oligochaeta	1	8							1		

6.8. 24.10.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Zelandobius	5				4				5	1	
Deleatidium	8			3			5	5	3		5
Hudsonema	6			10							
Hydrobiosis	5						30				
Oxyethira	2		8	10	10	40	10	30	10	30	10
Psilochorema	8									30	
Triplectides	5				1		30				3
Potamopyrgus	4	40	50	120	25	10	30	40	50	40	35
Physa	3	15			5	20	4			10	
Sphaeriidae	3		10				2		10		
Elmidae	6						15				
Chironomidae	1	60	80	200	25	150	80	25	75	80	80
Austrosimulium	3									10	
Tanypodinae	5	10	20								20
Ostracoda	3	25		25	10		15			30	
Amphipoda	5				20		40		10	25	40
Copepoda	5			25							
Oligochaeta	1	3	10	5	3	3	2	4	5	25	
Nematoda	3	2									

6.9. 23.11.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Zelandobius	5				4	1			5		
Deleatidium	8			10	5		5	5			
Hudsonema	6			25							
Hydrobiosis	5						30				
Oxyethira	2		11	10	7	100	20	30	20	100	30
Psilochorema	8				3					50	
Pycnocentrodes	5			10			5				
Triplectides	5						30				
Potamopyrgus	4	50	70	200	40		30	40	50	40	30
Lymnae	3							8			
Physa	3	10			5	20	20		5	20	
Sphaeriidae	3		15				3		10		
Elmidae	6					1	20				
Chironomidae	1	70	66	300		300	400	25	200	400	150
Austrosimulium	3			20						10	
Tanypodinae	5	10	30			50					20
Ostracoda	3	33		25	25		20			30	
Amphipoda	5			200	50		50				40
Copepoda	5			25							
Oligochaeta	1	3	10	5	5	3	3	4	30	100	3
Nematoda	3	2			5						

6.10. 3.12.2013

Taxa/Site	Taxon Score	1	2	3	4	5	6	7	8	9	10
Zelandobius	5				3	1			15	5	
Deleatidium	8			8	7		10	5	3		
Hudsonema	6			20							
Hydrobiosis	5						20				
Oxyethira	2		10	8	6	80	15	20	15	8	8
Psilochorema	8				3					15	2
Pycnocentrodes	5			8							
Triplectides	5						25				
Potamopyrgus	4	60	60	125	33	4	40	50	40	30	15
Lymnae	3							8			
Physa	3	15	2		5	15	20		3	15	
Sphaeriidae	3		15				15		21		
Elmidae	6						15				
Chironomidae	1	60	66	200		250	300	45	150	50	30
Austrosimulium	3			20							
Tanypodinae	5	8	25			50					
Ostracoda	3	20		30	25		25			40	
Amphipoda	5			200	60		50				30
Copepoda	5			25							
Oligochaeta	1	4	8	7	5	5		3	20	7	
Nematoda (horsehair)	3	2			7				15		